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Soil Processes Related to Marsh Loss in Coastal Louisiana.

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SOIL PROCESSES RELATED TO MARSH LOSS IN COASTAL LOUISIANA

A Dissertation

Submitted to the Graduate Faculty of the
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in partial fulfillment of the
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in

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by
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I have long considered myself one of the luckiest people in the world and this dissertation proves it. I am standing on the shoulders of giants such as Dr. William H. Patrick, Jr., Dr. Ron D. DeLaune, Dr. Robert H. Chabreck, Dr. Harry H. Roberts, Dr. S. Reza Pezeshki, and Dr. Paul H. Templet. Graduate students John Callaway and Keith Fessel often helped with field work and frequently listened to me ramble on about these studies on the long drives to and from field work.

My wife provided love and support throughout these years, and I must also thank my parents for their many years of love and support. Without these people, I cannot imagine where or what I would be.

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ABSTRACT

Marsh loss in Louisiana (~8,000 ha/yr) is attributed primarily to salt-water intrusion, flooding, and sediment starvation. Soil processes were studied to complement previous work identifying causes rather than mechanisms.

Relationships between soil mineral and organic matter were examined to better understand accretion, which counters subsidence and sea-level rise. More soil volume was occupied by organic matter than mineral matter in fresh, intermediate, and brackish soil. Estimated mineral sediment requirements for accretion were 6.7 times greater in salt marsh than in fresh marsh. Sediment requirements probably increase seaward because of sulfate in seawater, reduction of sulfate to toxic sulfides in soil, and the interaction between soil iron and soil sulfides. Decomposition rates were studied because they varied among marsh types. Decomposition was greatest in fresh, and least in brackish soil at all drainage depths tested. Thus, different decomposition rates were attributed to the different plant species that create soil organic matter rather than different field hydrological conditions.

Relationships among vegetation, soil, and marsh loss were studied in a deteriorating marsh near Lake Barre. Vegetation was stressed, which was attributed primarily to flooding. However, inadequate soil mineral matter also apparently prevented salt marsh from establishing in brackish areas experiencing salt-water intrusion, which suggested that the sediment regime was not changing although the salinity regime was changing. Excessive flooding resulted from inadequate accretion but accretion depended directly on peat accumulation, which indicated a positive feedback loop of inadequate plant production, inadequate accretion, and excessive flooding culminating in

plant mortality and marsh loss. The hypothesis that marsh loss always results from plant stress and subsequent feedbacks between excessive flooding and inadequate accretion was tested at Marsh Island. Pilot and follow-up studies indicated that marsh loss there occurred in healthy marsh via erosion in the marsh interior. Thus, increasing sedimentation, or reducing flooding or salt-water intrusion will not reduce all marsh loss because all marsh loss is not associated with plant stress. Additional study is needed to determine if landscape patterns are related to marsh loss processes.

SECTION: I PROLOGUE

CHAPTER 1

INTRODUCTION TO LOUISIANA MARSH LOSS

In the United States, conversion of marsh to open water causes the loss of valuable wildlife and fisheries habitat in the coastal marshes of Delaware Bay (Phillips, 1986), Chesapeake Bay (Kearney and Stevenson, 1991), North Carolina (Hackney and Cleary, 1987), Texas (Morton and Paine, 1990), and Louisiana (Gagliano et al., 1981). Marsh loss is particularly severe in Louisiana. An average of 10,847 ha of marsh converted to open water each year between 1955 and 1978 (Gagliano et al. 1981), but the rate recently slowed to 7,954 ha/yr between 1974-1983 (the last years for which data are analyzed (Britsch and Kemp 1991). In popular units, the rate has slowed from an acre lost every 20 minutes to an acre lost every half hour. It seems that marsh loss in coastal Louisiana was initially considered a loss for individual land owners, later a loss for the state, and is currently considered a national loss. The greater value now placed on these wetlands results partly from increased national environmental awareness, but primarily from an appreciation of the importance of these wetlands to the nation's navigation and fisheries needs.

The 10 million hectares of saline and non-saline marsh that parallel Louisiana's coastline are a dynamic landscape. Sediments deposited by the Mississippi River filled the shallow margins of the Gulf of Mexico and built these marshes over the last several thousand years (Coleman 1988). But this building was not a simple gradual gain of wetlands. Instead, the river built extensive delta lobes and then abandoned them when they became hydraulically inefficient. The marshes in the abandoned delta lobes

deteriorated as the river built a new delta in another location. Frequently, the river built new deltas on top of the sunken remains of former deltas. These delta lobes occupy the eastern half of the coastal zone, which is termed the Deltaic Plain. Marshes were also built in the western half of the coastal zone, which is called the Chenier Plain, but by another process. Marshes were built by mud flat accumulation when the active delta lobe was nearby. When the active delta lobe was far to the east, these marshes eroded. Today, significant delta building does not occur in the Deltaic Plain because the river is retained in a channel that keeps the active delta at the edge of the continental shelf where water is too deep. Significant marsh building does not occur in the Chenier Plain because this delta is too far to the east. Some marsh building occurs because of the active delta at the mouth of the Atchafalaya river, but that is limited because only 30% of the volume of the Mississippi is allowed to flow this way. However, the natural marsh destruction processes continue and have been accelerated by society's methods for meeting navigation, energy, and flood control requirements.

Accurate identification of the causes and mechanisms of marsh loss is required if mitigation efforts are to be effective. Marsh loss in Louisiana is perceived to result primarily from human made hydrologic changes, and the resulting stress on marsh vegetation and subsequent plant dieback and pond formation is hypothesized to be the primary mechanism of marsh loss (Turner, 1990a). The fundamental causes are a lack of spring flooding by the Mississippi River and a network of canals in the marsh. This causes several types of problems at the local scale: inadequate marsh vertical accretion resulting from rapid submergence and sediment starvation, (DeLaune et al., 1983a, Mendelssohn et al., 1983, Templet and Meyer-Arendt, 1988),

interruption of marsh drainage by spoil banks associated with the vast canal network (Mendelssohn et al., 1983, Turner, 1990a), and salt water intrusion into brackish and fresh marsh resulting from the canal network and lack of spring flooding (Mendelssohn et al., 1983, Sasser et al., 1986). These causes are quite different, but all are believed to lead to the same mechanism of marsh loss: plant stress and subsequent plant dieback. Important stresses in Louisiana are believed to be (1) salinity stress resulting from salt water intrusion and (2) waterlogging stress resulting from impoundment or insufficient elevation relative to mean water levels. In addition to the human caused flooding and saltwater intrusion, natural flooding and saltwater intrusion occur in abandoned delta lobes.

Mineral sediments are considered important because of the need for marsh vertical accretion. Marsh vertical accretion is the gradual formation of new marsh soil on the marsh surface, and it occurs in coastal marshes throughout the world (Mitsch and Gosselink 1986:178-181). If it did not occur, marsh vegetation would soon drown because sea level has been rising and will continue to rise. Marshes in Louisiana have to accrete faster than almost all other marshes because subsidence in Louisiana causes marshes to sink roughly 5 times faster than would occur otherwise (from data in Peltier and Tushingham 1989, and Penland and Ramsey 1990). This subsidence is also related to the delta lobe cycle of the river because much of Louisiana sits on river sediments that are still dewatering and compacting. If it were not for marsh vertical accretion, then marshes in abandoned delta lobes would deteriorate in decades rather than centuries.

The traditional view of marsh vertical accretion is that it is controlled by a negative feedback loop that maintains the marsh surface at the same

elevation as rising water levels (Mitsch and Gosselink 1986:187-181). When the marsh surface loses elevation relative to water levels, the marsh floods more often. This increases tidal delivery of mineral sediments and decreases the decomposition of organic matter. This would cause mineral and organic matter to accumulate on the marsh, raise the elevation of the marsh surface, and cause the marsh to flood less often. This in turn would prevent additional soil formation by reducing mineral sedimentation and increasing the decomposition of organic matter. In this way, the marsh surface would always be within a few centimeters of mean water levels; never too low and never too high for marsh vegetation to thrive.

Marsh loss is therefore related to many important soil characteristics and processes. Soil chemistry, nutrient content, and drainage are all related and largely determine the health of the vegetation considered so important at holding these fragile soils together. These characteristics depend in turn upon related processes such as vertical accretion, mineral sedimentation, belowground plant production, and soil organic matter decomposition. Many of these processes have been the subject of previous research, but they have not been examined in context with one another and with the goal of understanding how they relate to marsh loss in Louisiana. This dissertation documents a series of studies undertaken with the intent of increasing understanding of marsh loss in Louisiana by increasing understanding of the relationships among the fundamental soil processes involved. In addition to adding to the body of knowledge, this information might contribute to better management of Louisiana marshes.

CHAPTER 2

DESCRIPTION OF LOUISIANA COASTAL MARSHES

Extensive fresh, intermediate, brackish, and saline marshes occur in coastal Louisiana adjacent to the Gulf of Mexico. Sediments deposited by the Mississippi River filled the margins of the Gulf of Mexico and built these marshes over the last several thousand years (Frazier 1967). These marshes received periodic inputs of sediments and freshwater from the Mississippi River until the early 1900's, but they are now isolated from direct river flooding (Frazier 1967). Even though there is no river discharge into these marshes, extensive non-saline marshes still exist where water exchange with the Gulf of Mexico is restricted. This is because average rainfall (162 cm/yr) is greater than average evapotranspiration (102 cm/yr) in southeast Louisiana (Newton 1972). This results in extensive bands of fresh, intermediate, brackish, and saline marsh that parallel Louisiana's coastline. The distance between the northern edge of the fresh marshes and the southern edge of the saline marshes varies from 24 to 80 km (Chabreck and Linscombe 1982). *Spartina patens* (Aiton) Muhl., which dominates brackish and intermediate marshes, is the most important plant species overall, and is almost twice as common as the second most important species, *Spartina alterniflora* Loisel, which dominates the saline marshes (Chabreck 1970). Winds easily dominate the small, 0.3 m, lunar tides. Persistent north winds associated with winter storm passage completely drain tidally influenced water bodies (Chabreck and Hoffpauir, 1962). The climate is humid subtropical, with 152-163 cm of precipitation each year and a 300-310 day growing season (Newton, 1972).

Louisiana's microtidal coastal marshes are extremely flat and do not exhibit the high marsh/low marsh characteristics of many tidal marshes (e.g., Daiber 1986). Instead, *S. alterniflora* marsh gradually gives way to *S. patens* marsh over a distance of several km with no great difference between them in elevation. Brackish and saline marshes do differ in flooding however. *S. patens* marsh in Louisiana typically floods 120 times per year, and is flooded 43% of the time (Sasser 1977). *S. alterniflora* marshes in Louisiana are typically flooded 55% of the time with approximately 190 flooding events per year (Sasser 1977). In recent years, there has been an inland migration of saline marsh into formerly brackish marsh areas (Chabreck and Linscombe 1982). Some of this may result from the canals constructed throughout the marshes that increase water exchange between interior marshes and the Gulf of Mexico, but much likely results from the natural cycle of delta building and delta decay (Coleman and Gagliano 1967, Morgan 1967).

These are peat marsh soils as opposed to the more mineral deposits of the southeast Atlantic coast of the United States or Great Britain. In this respect they are more similar to the soils of the New England marshes of the United States. Mineral matter occupies less volume than organic matter in Louisiana fresh, intermediate, and brackish soils, and generally >85% of soil volume is pore space (Nyman et al. 1991). Nonetheless, mineral matter may indirectly control vertical accretion in Louisiana as well as in New England marshes because it is as a source of plant nutrients (Bricker-Urso et al. 1989).

Coastal marshes in Louisiana have experienced substantial net loss within the last 30 years (Gagliano and van Beek 1970, Adams et al. 1976, Turner and Cahoon 1987, Britsch and Kemp 1991). These marshes have always experienced wetland loss, but until recently, loss in one area was more

than offset by natural marsh creation in another area as a result of channel switching and sediment deposition by the Mississippi River. But the Mississippi River no longer switches channels because of flood control and navigation practices, and it has long since prograded to the edge of the continental shelf where water is too deep for building new land. Natural marsh creation in coastal Louisiana is now occurring only at the mouth of the Atchafalaya River and adjacent Chenier Plain, and is inadequate to offset marsh loss.

Marsh loss occurs in two landscape patterns in Louisiana (Leibowitz and Hill 1987). "Hotspot" is a term coined by Leibowitz and Hill (1987) to describe areas of high marsh loss rates embedded within areas of internal fragmentation where loss rates are much lower. Leibowitz and Hill (1987) found that although hotspots accounted for only 12% of all marsh in their study areas, they accounted for 43% of all marsh loss.

Measurements of vertical marsh accretion rates indicate that many deteriorating coastal Louisiana marshes are not accreting at rates sufficient to maintain them in the intertidal zone (DeLaune et al. 1978, DeLaune et al. 1983a, Hatton et al. 1983, DeLaune et al. 1986a). Furthermore, canals have directly destroyed 4-5% of the wetland area in Louisiana, and altered the natural hydrological conditions as well (Turner and Cahoon, 1987). Wetland vegetation, which is important in the structure of marsh soils, is sensitive to hydrological conditions; but the relative importance of subsidence and altered hydrological conditions to wetland loss is unknown. The rate of marsh loss varies 10 fold among different areas in the state (Britsch and Kemp 1991).

Inland marsh soil contains less mineral matter than streamside marsh soil in all 4 marsh types (Hatton et al. 1983), and as a result of elevational

differences, saline marsh vegetation is less robust in inland marshes than in streamside marshes (DeLaune et al. 1983b, Mendelssohn and McKee 1988, Pezeshki et al. 1988). The majority of wetland loss in coastal Louisiana, 70% to 90%, has occurred in inland marshes (Turner and Cahoon, 1987).

SECTION II: MARSH SOIL

CHAPTER 3

MINERAL AND ORGANIC MATTER RELATIONSHIPS IN MARSH SOIL

INTRODUCTION

Global sea level rise during the last 4,000 years averaged 0.08 cm/yr (Redfield 1967), and marsh elevation was maintained by vertical accretion of mineral and organic matter. But global sea level rise is currently estimated at 0.2 cm/yr (Peltier and Tushingham 1989), and may reach 1.0 cm/yr by 2050. Submergence of many coastal marshes may result, causing substantial wetland loss (EPA 1988). Vertical accretion of coastal marshes must counter not only global sea level rise, but also subsidence. Subsidence is greater in the Mississippi Deltaic Plain than at any other location on the northern Gulf of Mexico coast (Penland and Ramsey 1990). Subsidence is greatest on the coast and decreases inland (Kolb and Van Lopik 1958), but there are few estimates of subsidence rates. On the Louisiana coast, subsidence and global sea level rise combine to produce submergence rates up to 1.11 cm/yr (Penland and Ramsey 1990).

Knowledge of organic and mineral matter relationships in marsh soils of the Louisiana Deltaic Plain is limited. The purpose of this study was to examine mineral-organic matter relationships in the different marsh types found in the Mississippi Deltaic Plain; and from this information, to initiate estimates of sediment requirements within these marshes in relation to various submergence rates. Furthermore, the relationship between vertical accretion and incorporation of mineral and organic matter into these soils was also examined. An understanding of these relationships may be useful in efforts to mitigate coastal wetland loss in Louisiana and other areas.

METHODS

Soil bulk density and percent organic matter from 30 cores from 13 sites throughout the Mississippi Deltaic Plain and from published data (Hatton et al. 1983) from an additional 4 sites, 1 composite profile (n=7 cores from each site, were analyzed. The cores were from 38-50 cm in depth and represented sites from which samples had been taken for ^{137}Cs dating (Figure 1). Published vertical accretion measurements were from 69 cores from the same 17 sites (DeLaune et al. 1986b). In this section, active will be used to denote Active Delta Zone marshes, but to reduce wordiness, the term inactive will be dropped when referring to marsh types from the Inactive Delta Zone when possible. Only data from inland marsh areas were used in these analyses, and only active intermediate marshes were not sampled. All active marsh cores were taken from fresh marsh near the mouth of the Atchafalaya River.

Cores were collected with a 15 cm diameter, thin walled, sharpened coring tube as described by DeLaune et al. (1983a), with the exception that some cores were sectioned into 2-cm increments rather than 3-cm increments. Soil bulk density and percent organic and mineral matter were measured in each increment in some cores, and in every other increment in other cores. Soil bulk density was determined from the oven-dried weight of a known volume of wet soil. Percent mineral and organic matter by weight in oven dried soil was determined by loss on ignition as described by Ball (1964). Ignition at low temperature to determine organic matter is accurate for calcareous (Davies 1974) and non-calcareous soils (Ball 1964).

Mineral and organic matter relationships were examined on a volume basis because volume probably more closely reflects structural importance in

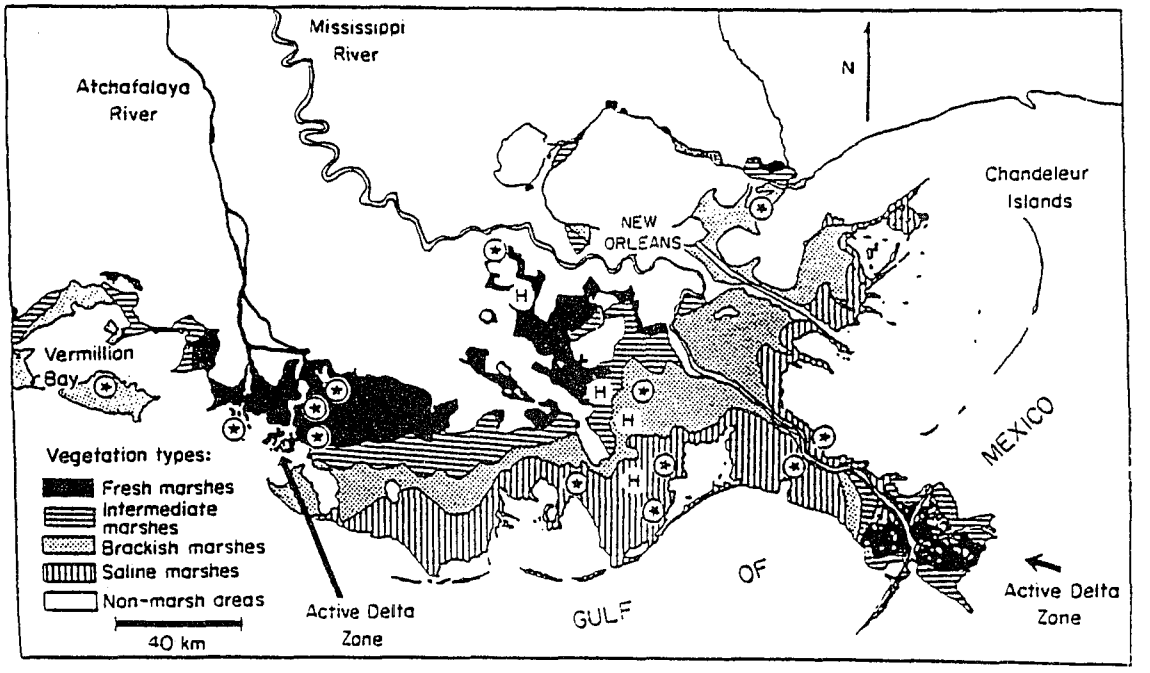

Figure 1. Approximate location of sites where soil was examined for mineral and organic matter relationships; stars represent individual core data, H's represent composite core data from Hatton et al. (1983).

marsh soil than weight. Particle density of mineral matter (2.61 g/cm3) and organic matter (1.14 g/cm3) (DeLaune et al. 1983a), and the measurements of soil bulk density and percent weight of organic and mineral matter were used to estimate the volume of the mineral and organic matter in soil increments. Volume not occupied by mineral or organic matter (pore space) was assumed to be occupied by water and gas. Volume profiles for each of the soil cores were created in this way. Composite volume profiles for each marsh type were created from the individual volume profiles.

The volume of mineral matter, organic matter, and water and gas in the upper 10 cm of soil was compared between Active and Inactive Delta Zones, and also among the marsh types of the Inactive Delta Zone, as weighted Randomized Block Design-Analysis of Variances (ANOVA) blocking on site, with Proc GLM of SAS (SAS Institute Inc., Cary N.C., USA 27512-800). The Waller-Duncan K-ratio t-test was used as a post ANOVA technique to determine which of the marsh types of the Inactive Delta Zone differed from one another. For these analyses, it was assumed that there was no relationship between depth and the volume of organic matter, mineral matter, or porosity in the upper 10 cm of soil. To test this assumption, depth was regressed against these variables within each marsh type as a weighted Randomized Block Design blocking on site, with Proc REG of SAS. An alpha level equal 0.0500 was chosen as the critical limit for all analyses.

In the statistical analyses, the assumption of independent and normally distributed error was suspect because these are percentage data near the limits of 0 and 100 (Steele and Torrie 1980, p. 233-237). Normality could not be tested directly with available methods because these are unbalanced data. Therefore the means of the residual error terms were

tested using Proc Univariate of SAS. The S-statistic, box plot and distribution of residuals were all considered before concluding normality. Several transformations were tried for each variable, and the log and square root transformations (Steele and Torrie 1980:234-235) were found to produce the most normally distributed error terms.

The weight of organic and mineral matter required to form 0.5, 1.0 and 1.5 vertical cm of marsh soil in the marsh types of the Inactive Delta Zone was calculated from the average soil bulk density and percent organic and mineral matter within the top 10 cm of soil. The weight of organic and mineral matter was used rather than volume because sediment availability and organic matter production are more likely to be reported on a weight basis. The weights of mineral and organic matter were plotted against vertical accretion, and the equation of the resulting line was calculated. The slope of the line is the organic and mineral matter requirements (g/m^2 per vertical cm) for soil formation in relation to submergence. Only the top 10 cm of soil were used because at some point in the column the soil begins to compact and dewater. Inclusion of soil below this point would overestimate mineral and organic matter, and underestimate water and gas, in the soil at the time of formation.

RESULTS

Regression analysis showed that there were no significant effects of depth on the percent volume of organic matter ($\underline{P} = 0.9575$), mineral matter ($\underline{P} = 0.1409$) or water and gas ($\underline{P} = 0.2649$) within the upper 10 cm of soil in any marsh type. Compaction must begin deeper in the soil profile.

ANOVA indicated that active fresh marsh soil differed from inactive fresh marsh soil with respect to the volumes of organic matter, mineral matter and water and gas. Active fresh marsh soil had more mineral ($\underline{P}$ = 0.0001) and organic matter ($\underline{P}$ = 0.0001) than inactive fresh marsh soil, hence porosity was lower ($\underline{P}$ = 0.0001). Organic matter ($\underline{P}$ = 0.0001), mineral matter ($\underline{P}$ = 0.0061), and water and gas ($\underline{P}$ = 0.0070) differed among sites, although replicate cores within sites did not differ (organic matter: $\underline{P}$ = 0.1756, mineral matter: $\underline{P}$ = 0.1845, water and gas: $\underline{P}$ = 0.1319).

Within the Inactive Delta Zone, bulk density increased from fresh (inland) to saline (seaward) marshes; whereas percent organic carbon by weight generally decreased (Table 1). However, intermediate marsh had higher percent organic carbon than fresh marsh. Porosity was greater than 85% throughout the upper 50 cm of soil in all but saline marsh (Figure 2). Among inactive marsh types, porosity in brackish marsh soil most resembled porosity in active fresh marsh soil (Figures 2-3). Bulk density and percent organic matter in brackish marsh were also more similar to active fresh marsh than were other marsh types of the Inactive Delta Zone.

ANOVA indicated differences in the volumes of organic matter, mineral matter, and water and gas among the marsh types within the Inactive Delta Zone. Post analyses comparisons indicated that intermediate marsh soil was similar to brackish and saline marsh soils with respect to organic matter, but more like fresh marsh soil with respect to mineral matter and water and gas. Saline marsh soils had more mineral matter and less water and gas than other marsh types. Organic matter ($\underline{P}$ = 0.0039), mineral matter ($\underline{P}$ = 0.0001), and water and gas ($\underline{P}$ = 0.0001) differed among sites within different marsh types. Replicate cores from the same site did not differ with respect to

Table 1. Bulk density (g/cm^3), % organic matter, vertical accretion (cm/yr), and volumes of organic matter, mineral matter, and water and gas in the upper 10 cm of soil from inland marshes of the Mississippi River Deltaic Plain.

	Active Delta	----------Inactive Delta----------			
	fresh	fresh	intermediate	brackish	saline
# of sites	3	2	1	4	7
# of cores	7	8	7	17	18
bulk density	0.14	0.07	0.08	0.16	0.24
(std. dev.)	0.05	0.03	0.05	0.07	0.11
% weight organic	42.8	38.6	56.6	36.8	27.6
(std. dev.)	10.3	11.6	12.9	10.5	13.9
vertical[b] accretion	0.86	0.67	0.64	0.72	0.72
(std. dev.)	0.11	0.02	0.38-1.06[a]	0.08	0.14
% organic volume	4.91	2.36	3.96	5.11	5.27
% mineral volume	3.18	1.63	1.33	4.03	6.89
% pore volume	91.91	96.02	94.70	90.86	87.84

[a] range

[b] vertical accretion was determined at the same site, but from additional cores as well, 69 cores in all

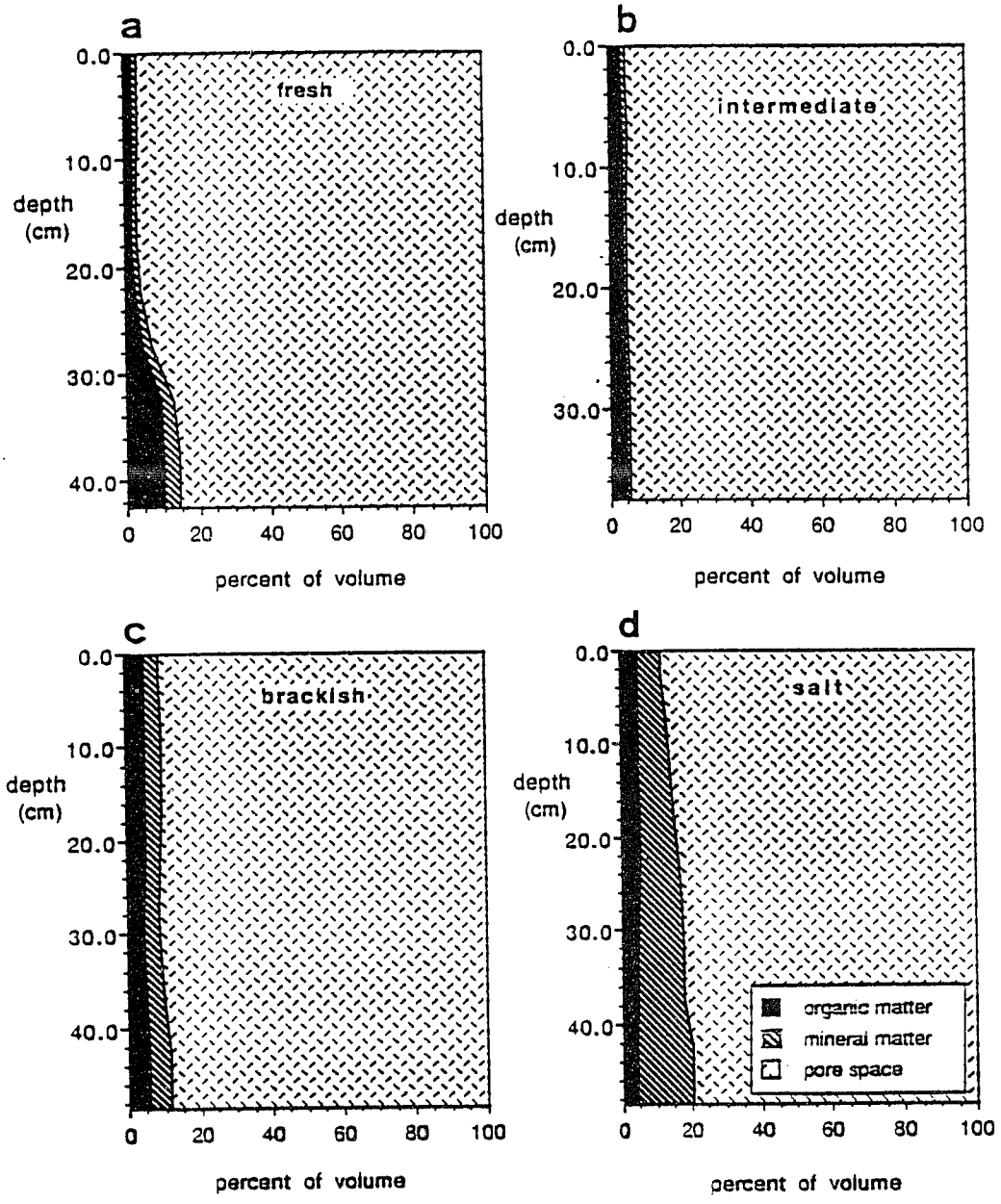

Figure 2. Volumes of mineral matter, organic matter, and pore space with depth in the different marsh types of the Inactive Mississippi River Deltaic Plain.

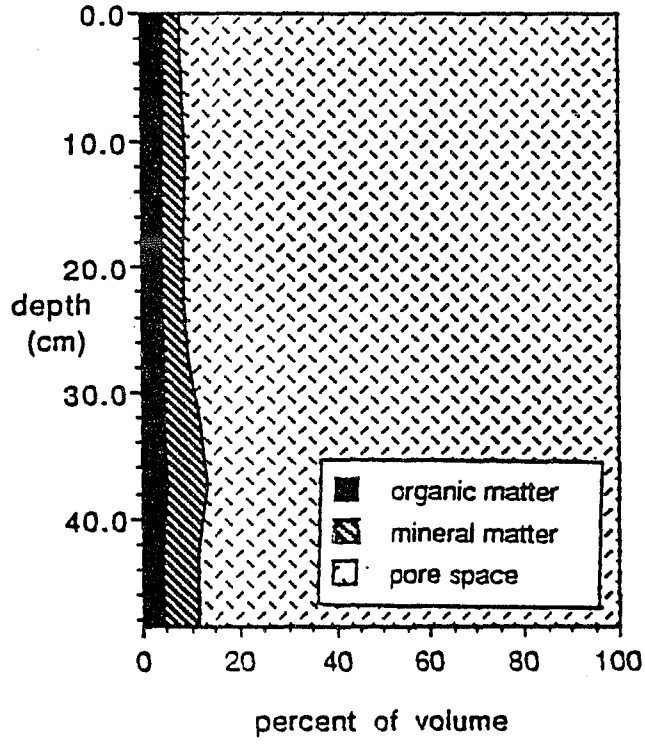

Figure 3. Volumes of mineral matter, organic matter, and pore space with depth in fresh marsh in an Active Delta Zone (Atchafalaya Delta Region).

organic matter ($\underline{P}$ = 0.2109) or porosity ($\underline{P}$ = 0.3346), but mineral matter did differ among some replication cores ($\underline{P}$ = 0.0071).

Within the upper 10 cm of Inactive Delta Zone marsh, the volume of organic matter increased from inactive fresh (inland) to saline (seaward) marsh, but for mineral matter this trend was interrupted in intermediate marsh (Table 1). Saline marsh contained 2.2 and 4.2 times more organic matter and mineral matter than inactive fresh marsh. Porosity differed 10% between inactive fresh and saline marsh, and generally decreased from fresh (inland) to saline (seaward) marshes, but this trend was also interrupted in intermediate marsh. On a volume basis, organic matter was more important than mineral matter in all but saline marsh soil. The ratio of organic matter to mineral matter was similar in active fresh (1.5:1), inactive fresh (1.4:1) and brackish marsh (1.3:1). In inactive intermediate marsh soil the organic matter to mineral matter ratio was 3.0:1 and in saline marsh soil it was 0.8:1.

Vertical accretion rates were greatest in the Active Delta Zone (Table 1). Within the Inactive Delta Zone, accretion rates in brackish and saline marsh were similar, and greater than in fresh and intermediate marsh (Table 1). Accretion rates were most variable in saline and active fresh marshes, presumably where mineral matter availability was greater (Table 1). On a weight basis, the requirements for mineral matter were greater than for organic matter in all marsh types (Figure 4).

DISCUSSION

Mineral Matter

Plant growth in these marshes is enhanced by mineral matter because mineral matter releases mineral nutrients such as calcium, potassium and

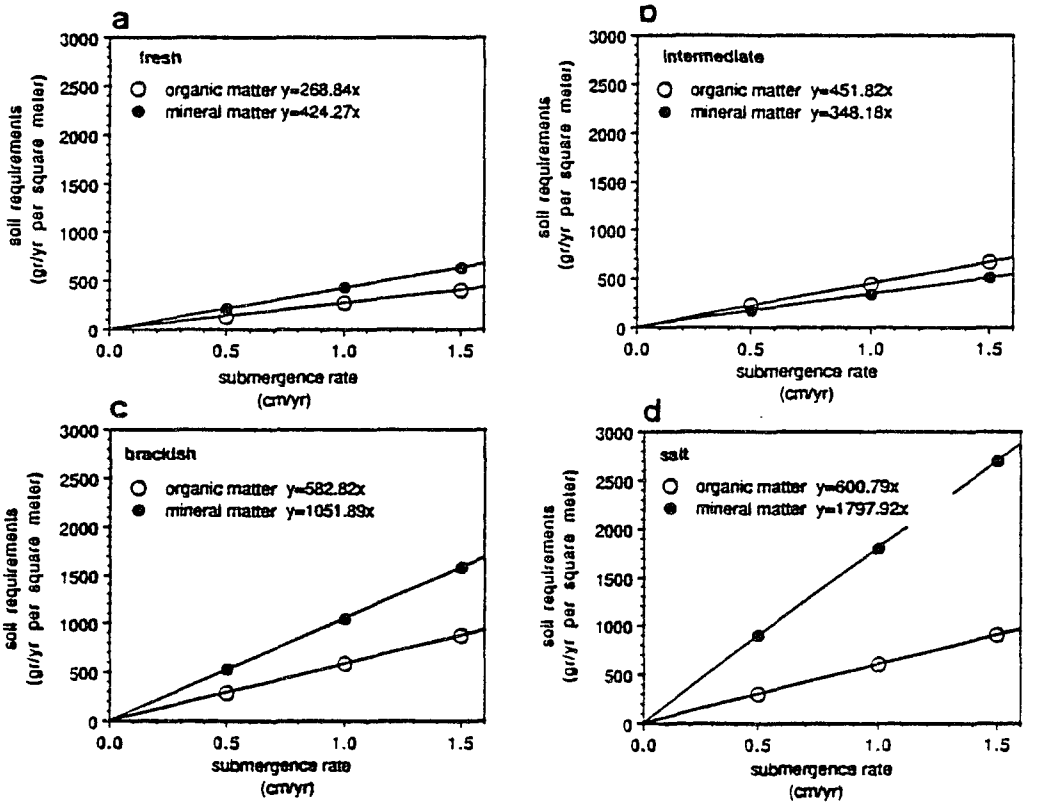

Figure 4. Mineral and organic matter accumulation requirements for marsh soil formation relative to submergence.

phosphorus, and under reduced conditions provides cation exchange and sorption sites for ortho-phosphate which restricts phosphate leaching (Patrick and Khalid 1974). Soil P and saline marsh vegetation productivity are positively related in North Carolina (Broome et al. 1975). End-of-season standing crop biomass of *Spartina alterniflora* is positively related to soil P, soil Fe, and soil bulk density in Louisiana saline marsh (DeLaune and Pezeshki 1988). King et al. (1982) reported a positive relationship between soil Fe concentrations and vegetation productivity in a Georgia marsh. Mineral matter has little affect on nitrogen, which is a major plant nutrient, and these marshes are nitrogen limited (Buresh et al. 1980, DeLaune et al. 1986c, Patrick and DeLaune 1976), and depend on the mineralization of organic matter for nitrogen.

Comparison between Delta Zones- Fresh marsh in the Active Delta Zone contained 2.1 times more mineral matter than fresh marsh in the Inactive Delta Zone. Mineral matter availability was undoubtedly greater in active fresh marsh water bodies than in inactive fresh marsh water bodies. Furthermore, the inactive fresh marsh sampled was floating marsh which is seldom covered by water, whereas active fresh marsh is regularly covered by flood waters. The combination of greater availability and delivery of mineral matter in active fresh marsh was most likely responsible for the greater volume of mineral matter in active fresh marsh soil.

Comparisons among Marsh Types- To some degree, mineral matter appeared to be incorporated into soil in proportion to availability. In fresh and intermediate marshes, where mineral matter is probably not as available, mineral matter contributed less than 2.0% to the volume of these soils. In the other marsh types, mineral matter contributed 4.0% to 6.9% of

the volume. The low volume of mineral matter in inactive fresh marsh soils was not a result of marsh type; active fresh marsh, where mineral matter is more available, incorporated more mineral matter.

Organic Matter

A given volume of organic matter probably contributes more to soil structure than a similar volume of mineral matter because soil organic matter is either living root mass, or is partially decomposed plant parts which retain some structural integrity. Thus soil organic matter forms interlocking networks that particulate mineral matter alone cannot. This network was in all soil increments, but appeared strongest from a few centimeters below the surface down to 10-20 cm below the surface, i.e. within the living root zone. There may be an optimal ratio of organic matter to mineral matter to pore space which would require the least amount of organic and mineral matter to produce structurally strong soils and promote vigorous plant growth.

Comparison between Delta Zones- Fresh marsh in the Active Delta Zone contained 2.0 times more organic matter than fresh marsh in the Inactive Delta Zone. The reason for the greater volume of organic matter in active fresh marsh soil than in inactive fresh marsh soil is not clear, but perhaps the annual influx of mineral matter in active fresh marsh supplied phosphate, a limiting nutrient in inactive Louisiana fresh marsh vegetation (Mitsch and Gosselink 1986, p. 267-268). Unlike inactive fresh marsh vegetation, vegetation in Louisiana saline marsh contains more mineral matter and is not phosphate limited (Buresh et al. 1980, Patrick and DeLaune 1976), probably as a result of the relatively unweathered sediments and to an abundance of sorption sites for phosphate.

Comparisons among Marsh Types- Within the Inactive Delta Zone, fresh marsh soils were different from other marsh soils with respect to the volume of organic matter. Net aerial primary production has been estimated at 1,960 g organic matter/yr m^2 in inactive fresh marshes (Sasser and Gosselink 1984) where the volume of soil organic matter was lowest, and 1,540 g organic matter/yr m^2 in inland saline marshes (Kirby and Gosselink 1976) where the volume of soil organic matter was highest. Although these are estimates of above-ground productivity, and soil organic matter is probably mostly derived from below-ground biomass, this suggested that the availability of organic matter for soil formation was similar in fresh and saline marsh. Intermediate marsh soil contained low volumes of mineral matter similar to fresh marsh, but more organic matter than inactive fresh marsh, which suggested that the low volume of mineral matter in fresh marsh soil did not limit incorporation of organic matter. It therefore appeared that neither the availability of organic matter or mineral matter limited incorporation of soil organic matter in inactive fresh marsh, and it could not be determined from these data why fresh marsh soil contained less organic matter than other Inactive Delta Zone marsh types.

Intermediate marsh soil had low volumes of mineral matter similar to inactive fresh marsh, but high volumes of organic matter similar to brackish and saline marsh. Although this marsh type may be grouped with brackish marsh because both are often dominated by *Spartina patens*, these data suggest that these marshes deserve separate classification. The volume of organic matter was greatest in saline marsh, but mineral matter occupied more volume; this was the only marsh type in which mineral matter occupied substantially more volume than organic matter.

Vertical Accretion

Comparison between Delta Zones- Only in the Active Delta Zone, where mineral matter availability was greatest, did accretion rates exceed 0.70 cm/yr. This was most likely a result of the high sediment load of these waters. These active fresh marsh soils incorporated half as much mineral matter as saline marsh but vertically accreted faster, and have expanded in recent years, whereas salt marshes have experienced substantial loss in recent years (Adams et al. 1976).

Comparisons among Marsh Types- Mineral matter availability and subsidence rates are low in inactive fresh and intermediate marsh, so which was limiting vertical accretion rates in these marshes could not be determined. At low rates of submergence, vertical accretion is limited by the rate of submergence (see Mitsch and Gosselink 1986, p. 178-181). If this were not the case, then marshes would accrete until they were upland habitats.

Rates of vertical accretion in brackish and saline marshes were similar, but saline marshes contained greater volumes of mineral matter. The dominant saline marsh plant, *Spartina alterniflora*, does not grow in soils with bulk density much less than 0.20 g/cm^3 (DeLaune et al. 1979). *Spartina patens*, the dominant plant in brackish and intermediate marsh, and *Panicum hemitomon*, the dominant fresh marsh plant, occupy soils with lower density where permitted by low salinity. This extra need for mineral matter by saline marsh vegetation was probably required to precipitate sulfide. Sulfide is toxic to marsh vegetation (Pezeshki and DeLaune 1988) and is formed from sulfate under reduced conditions; it appears in the soil solution after all available iron has precipitated with sulfides. The primary source of sulfate in

coastal marshes is sea water. This may explain why saline marsh required almost twice as much mineral matter as inactive brackish marsh, but accreted at a similar rate. Allocation of limited mineral matter in a way that increases the rate of vertical accretion at the expense of soil bulk density may lead to vegetation die-back in saline marsh. With plant mortality, the structurally important root network would be lost and soil disintegration may follow. In this way, saline marsh soil formed with a density less than 0.20 g/cm^3 would be eroded away.

Soil Formation Requirements

Soil formation requirements were not estimated for Active Delta Zone marsh because the organic and mineral matter contents of those marshes probably more reflected fluctuations in the excess of mineral matter availability than amounts required for soil formation. In the Inactive Delta Zone however, the organic and mineral matter contents in the near surface are more typical of the amounts required for soil formation in submerging coastal marshes. In these marsh types, mineral matter requirements were greater than organic matter requirements on a weight basis (Figure 4). Saline marsh required almost twice as much mineral matter as brackish marsh. Inactive fresh and intermediate marsh required less than half as much mineral matter as brackish marsh.

Comparison of mineral matter availability to the estimates of mineral matter requirements could indicate if mineral matter was limiting vertical accretion at a given submergence rate, but I am unaware of mineral matter availability estimates. Estimates of mineral matter availability should be made in inland marsh rather than merely of the sediment load in bayous and ponds because much of the mineral matter in water bodies is deposited on

streamside marsh and is thus unavailable to inland marsh. Also, the actual delivery of mineral matter to inland marsh is probably a function of the settling properties of the suspended particles and the standing time of the water on the marsh, as well as the sediment load of the flooding waters.

Future efforts to determine if production of organic matter is limiting vertical accretion should compare organic matter availability to soil formation requirements. Availability is a result of the interactions of above and below-ground productivity with detrital export, fire and herbivory. Estimates of below ground productivity and the effects of herbivory and fire are lacking in these marshes. Demand is a combination of oxidation of soil organic matter by soil microbes to CO_2 and CH_4, and soil formation requirements. Oxidation of soil organic matter to CO_2 and CH_4 have been estimated in inactive fresh, brackish and saline marshes (Smith et al. 1983, DeLaune et al. 1983c), but are lacking for active fresh and intermediate marshes, and inactive intermediate marshes. Replacement of soil organic matter oxidized by microbial respiration increases the organic matter requirements (Figure 5). These figures and equations represent our current best estimate of organic and mineral matter requirements of the marsh types in the Inactive Delta Zone of the Mississippi River Deltaic Plain. At submergence rates of 1.0 cm/yr, organic matter requirements approach estimates of net aerial primary production, although below-ground productivity probably contributes the bulk of organic matter. Estimates of below-ground productivity in these marshes are lacking but range from equal to above ground productivity in a Mississippi brackish *Juncus roemerianus* marsh (de la Cruz et al. 1977), to 8 times net aerial primary productivity in a Massachusetts inland saline *Spartina alterniflora* marsh (Valiela et al. 1976).

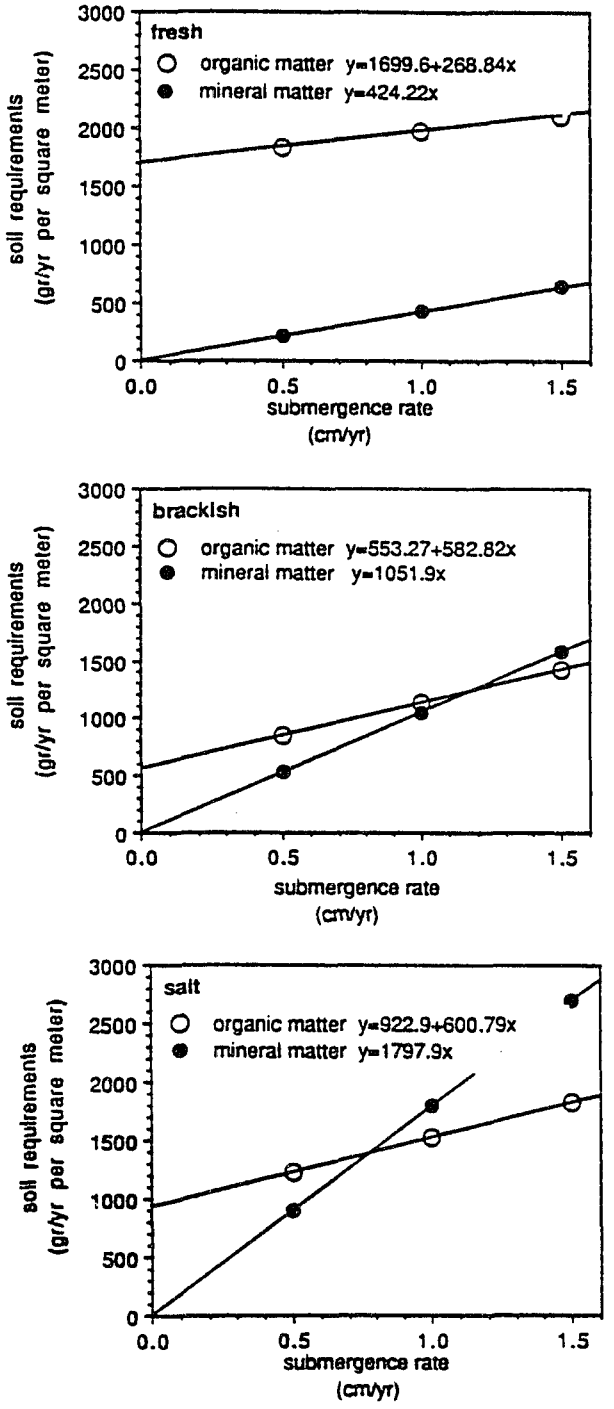

Figure 5. Mineral and organic matter inputs required for marsh soil formation in different marsh types relative to submergence. Required organic matter inputs differ from required organic matter accumulation because of soil organic matter decomposition.

CONCLUSIONS

Saline marsh soil required 1.7 times as much mineral matter as brackish marsh soil to vertically accrete at similar rates, possibly as a result of soil bulk density requirements of the dominant saline marsh plant, *Spartina alterniflora*. Vertical accretion rates were highest in the Active Delta Zone, probably as a result of increased mineral matter availability and delivery. Current, best estimates of the combination of mineral and organic matter required to maintain marsh surface-water level relationship are:

Marsh Type	Organic Matter	Mineral Matter
fresh	y=1,700 + 269x	y=424x
brackish	y=553 + 583x	y=1,052x
saline	y=923 + 601x	y=1,798x

where x = the rate of submergence (cm/yr),

and y = the amount required (g m^{-2} yr^{-1}).

CHAPTER 4
SOIL ORGANIC MATTER DECOMPOSITION IN FRESH, BRACKISH, AND SALINE MARSHES

INTRODUCTION

In the previous study, it was found that the different marsh types had different soil mineral matter requirements. The greater availability of sulfate in seawater, the reduction of sulfate to toxic sulfides by soil bacteria, and the role of soil iron in buffering sulfides are the most likely reason that mineral matter requirements increase seaward. Organic matter requirements differed because of different soil respiration rates, but reasons for the different respiration rates were unknown. These respiration rates may be important in these coastal marshes. Consumption of organic matter and the subsequent production of CO_2 contributes to the erosion of peat (Stewart and Wheatley 1990). This chapter documents an investigation of the factors controlling soil respiration rates in the different marsh types.

Although there may be no reason to expect CO_2 emissions to vary among marsh types (see Odum 1988), Smith et al. (1983) reported that CO_2 emissions were highest in fresh marsh, and lowest in brackish marshes in Louisiana. The differences reported by Smith et al. (1983) might be partially related to differences in hydrological conditions among marsh types. Tidal range is greater in saline marshes than in brackish marshes, and slight to nonexistent in fresh marshes (Bahr and Hebrard 1976); thus one purpose of this study was to compare the effects of hydrological conditions on CO_2 emissions among the fresh, brackish, and saline marshes of the Mississippi River Deltaic Plain. A second purpose was to examine the effects of

hydrological conditions on soil Eh. Changes in marsh soil water chemistry coincide with changes in soil Eh (Feijtel et al. 1988), and soil Eh has been used as an indicator of waterlogging stress on saline, brackish, and fresh marsh vegetation (DeLaune et al. 1983a, Burdick et al. 1989, McKee and Mendelssohn 1989).

METHODS

Marsh Soil Cores

Rates of CO_2 emission, soil Eh, and soil temperature were measured in 27 intact marsh cores (9 each from 3 marsh types) in which hydrological conditions were controlled. The cores were 13.5 cm in diameter and 25-30 cm long, and were collected from fresh, brackish, and saline marshes within Barataria Basin, Louisiana (Figure. 1). Soils at these sites are extensive peats and have been described by Hatton et al. (1983) and Smith et al. (1983); all are highly organic with low bulk density (Table 2). All cores were taken from within 10-m^2 areas about 10 m from small bayous, except in the saline marsh area where cores were collected >20 m from a bayou to avoid the streamside zone. The fresh marsh site was dominated by *Panicum hemitomon*, the brackish site was dominated by *Spartina patens*, and the saline marsh site was dominated by *Spartina alterniflora*. Bayou water was also collected when the cores were collected.

A thin walled aluminum coring tube was used to collect cores, and cores were transferred to PVC storage tubes in the field. Maximum effort was made to minimize soil disturbance. Vegetation growing in cores was clipped at the soil surface. Cores were stored flooded for 2-6 weeks in the laboratory to allow the microbial communities to return to equilibrium. Vegetation in the

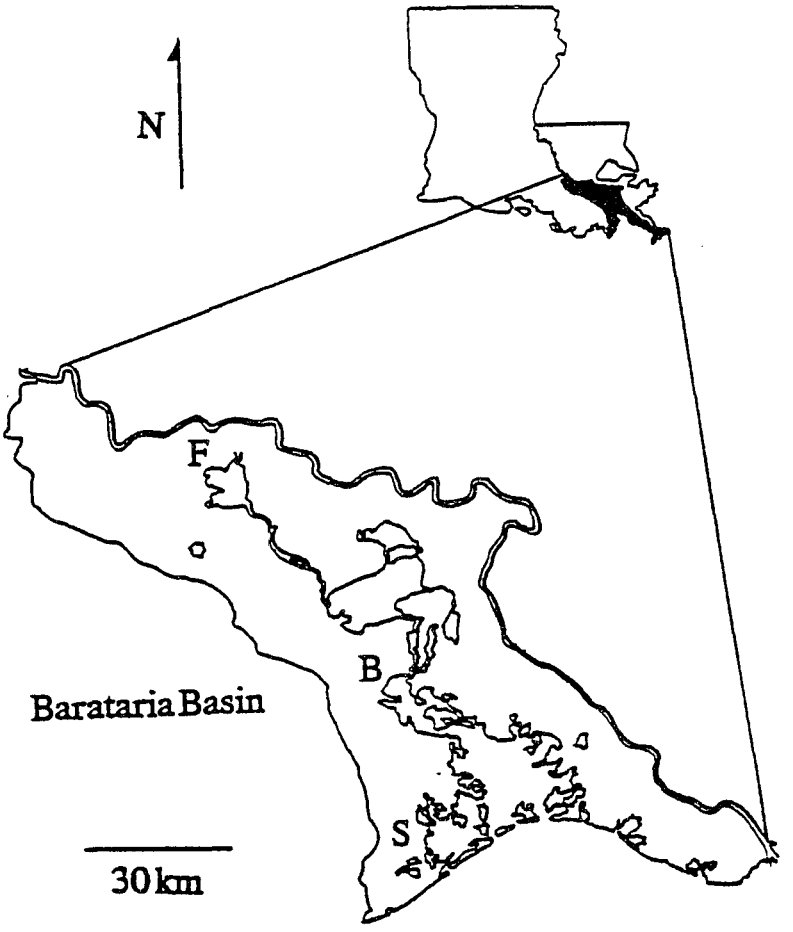

Fig. 6. Fresh (F), brackish (B), and saline (S) marsh sites where intact marsh cores were collected for study of soil organic matter decomposition in Barataria Basin, Louisiana.

Table 2. Soil characteristics at sites where cores were collected for study of CO_2 emission and soil Eh, data from Hatton et al. (1983), and Smith et al. (1983).

Soil Characteristic Site	Fresh Site	Brackish Site	Saline
bulk density (g/cm^3)a	0.09 ± 0.01	0.14 ± 0.01	0.29 ± 0.06
bulk density (g/cm^3)b	0.10 ± 0.04	0.13 ± 0.02	0.28 ± 0.06
organic matter (% dry weight)a	52 ± 6	42 ± 6	20 ± 3
organic matter density (g/cm3)a	0.047	0.059	0.058
organic C density (mg/g)b	263 ± 14	240 ± 32	86 ± 16
organic C accumulation (g m-2 yr-1)b	224	296	183
CO_2-C emissions (g C m-2 yr-1)b	618	180	418
vertical accretion (cm/yr)a	0.65	0.59	0.75
vertical accretion (cm/yr)b	0.85 ± 0.02	0.95 ± 0.05	0.76 ± 0.12

a Data from Hatton et al (1983)
b Data from Smith et al. (1983)

cores did not survive the storage period. Morris and Whiting (1986) concluded that diffusion of CO_2 through stems was substantial, but this pathway for CO_2 loss was not available in the cores used in this study, and this pathway and was not addressed.

Experimental Design

The water-table depth was manipulated by drilling a 0.87 cm hole in the storage tube at one of 4 depths relative to the soil surface: 0 (surface), -7.5 cm, -15 cm, and -22.5 cm. Fiberglass inserted into the drain holes reduced sediment losses during drainage, which took 5-20 min. Drain holes where corked to prevent air entry into the core when cores where drained, and to prevent water loss when flooded. Distilled water was added to the storage tubes as needed to prevent water evaporation from causing salinity increases. Bayou water was added as needed to increase water depths. Water-table depth in one saline core was accidentally set at -10.0 cm rather than -7.5 cm during one treatment.

Several measurements of soil Eh, soil temperature, water depth, and CO_2 emission were made on each core over a 2 week period of continuous flooding. Eight cores of the nine from each marsh type were then randomly assigned to either a static water level treatment or a tidal-like water level treatment. Cores assigned to the static water level treatment (n = 4 per marsh type) were then drained to one of the water-table depths for one week, and measurements of CO_2 emission, soil Eh, and soil temperature were made several times on each core, after allowing 2 d for the cores to equilibrate. The water-table depth was then changed, and the cores were kept drained for a second week, and the measurements were repeated. The

change in water-table depth was needed so that variation among cores could be estimated.

Water levels in cores assigned to the tidal-like water level treatment (n = 4 per marsh type) were manipulated on a 24 h cycle to mimic the effects of tidal drainage to the four different water-table depths. Each day, water was drained at approximately 0400 hours, and the cores were reflooded at approximately 1600 hours. Measurements of soil Eh, soil temperature, water depth, and CO_2 emission rates were made twice daily, once when the cores were drained, and again when the cores were flooded. Two days of tidal-like water levels were allowed to pass before CO_2 measurements were begun, and the treatment ran for another 6 d. Three days into the course of the experiment, the water-table depth was changed to allow estimation of variation among cores. Drained water was returned to a common storage reservoir and reused to prevent leaching of soil nutrients.

The remaining three cores, one core from each marsh type, were never drained and served as a test for effects of storage time. Soil Eh, soil temperature, water depth and rates of CO_2 emission were measured in these untreated cores from 34 to 241 d after they were collected (treated cores were measured from 34 to 169 d after collection). Unlike the continuously flooded treatment which was monitored for only a 2-3 week period prior to drainage treatments, measurements were made on the untreated cores anywhere from 34 to 241 d after collection.

Measurements

Soil Eh was measured in each core with permanently installed platinum tipped Eh electrodes (Faulkner et al. 1989) located at five depths (hereafter referred to as levels to avoid confusion with water-table depth): -1

cm, -5.0 cm, -7.5 cm, -15 cm, and -22.5 cm; 1,619 measurements of soil Eh were statistically analyzed.

A portable open gas exchange system (Analytical Development Co., model LCA-2, field analytical system) was used to measure rates of CO_2 emission. The system measures differences in CO_2 concentrations between air flowing into and out of a soil chamber, and consists of an infrared gas analyzer, ADC model LCA-2, and an air supply unit that regulates the flow of air through the system. The soil chamber fitted onto the storage tube without disturbing the soil, and the seal was made air-tight with several wraps of Parafilm (American Can Co.). Air flow through the soil chamber was usually maintained at 300 ml min^{-1}. Before CO_2 measurements began, a minimum of 20 minutes was allowed to pass after sealing the soil chamber because it was necessary to purge room air out of the soil chamber, and to allow the $[CO_2]$ in the soil chamber to equilibrate with the rate of CO_2 emission from the soil and the rate of air flow through the chamber. The concentration of CO_2 was then monitored every minute for another 20 minutes to insure that equilibrium between CO_2 emissions and flow rate had occurred. All observations consisted of the mean of at least 20 measurements, spaced 60 s apart; 341 such observations were statistically analyzed and used to compute treatment means. Compressed breathing air ranging from 80 to 500 ppm CO_2 between tanks was supplied to the soil chamber because CO_2 concentration in room air varied greatly and would have caused incorrect estimates of CO_2 emissions.

Preliminary investigations indicated that extremely large amounts of CO_2 were released as cores were drained, and that CO_2 emissions fell as cores were flooded. This was assumed to be a release of entrapped gases

upon drainage, rather than a true increase in emission, and entrapment of produced CO_2 within the soil, rather than from a true decrease in CO_2 production. For these reasons, CO_2 measurements were made at least 4 h after draining or flooding.

Statistical analyses

Cores were held in the laboratory for extended periods of time, and it was necessary to assume that soil Eh and CO_2 emissions were not affected by prolonged, flooded storage. To test this assumption, CO_2 emissions from untreated cores were analyzed as an ANOVA (alpha = 0.0500) testing for differences among cores, with regression of water depth, temperature, and time of storage. Post analyses comparisons among cores were made with Waller Duncans' k ratio t-test. Eh was similarly analyzed, but with the addition of soil level as a random effect split plot.

CO_2 emission data were analyzed as a 3X2 factorial, marsh type by drainage regime, with multiple regression of water-table depth and soil temperature to test for differences among marsh types and for a relationship with water-table depth. Marsh type and drainage regime were fixed effects, water depth and soil temperature were random effects. Interaction terms were tested using the core within marsh type variation as the error term; error terms for main effects varied because some random interaction terms were significant. Type III sums of squares and a critical limit of 0.0500 were used. Soil temperature interaction terms were excluded from analyses a priori, and other interaction terms were excluded from the model when appropriate; alpha = 0.2000.

The relationship between soil Eh, marsh type, hydrological condition, and soil level was also analyzed. Eh data were analyzed similar to the CO_2

data, but with the addition of soil level as a random effect, split plot. Main plot terms were tested as in the above analyses; split plot terms were tested with the proper error term as indicated by significant split plot interaction terms. Type III sum of squares and a critical limit of 0.0500 was used. Soil temperature interaction terms were excluded from analyses a priori, and other interaction terms were excluded from the model when appropriate; alpha = 0.2000.

RESULTS

Untreated Cores

There was no relationship between the length of time untreated cores were held in the laboratory, up to 241 d, and CO_2 emissions ($F = 0.73$, 1 and 47 df, $P = 0.3980$). CO_2 emissions varied among untreated cores ($F = 106.62$, 2 and 47 df, $P = 0.0001$), with water depth ($F = 8.53$, 1 and 47 df, $P = 0.0054$), and with soil temperature ($F = 21.13$, 1 and 47 df, $P = 0.0001$). Post ANOVA comparisons indicated that CO_2 emissions were greater in the fresh core (mean = 195 g C yr^{-1} m^{-2}, n = 14, SE = 8) than in the brackish (mean = 91 g C yr^{-1} m^{-2}, n = 28, SE = 6) or saline core (mean = 78 g C yr^{-1} m^{-2}, n = 12, SE = 5). CO_2 emission was negatively related to water depth, and positively related to soil temperature, which ranged from 16.0°C to 26.4°C.

There were no relationships between soil Eh in flooded untreated cores and the length of storage time ($F = 0.16$, 1 and 2 df, $P = 0.7242$), water depth ($F = 0.00$, 1 and 2 df, $P = 0.9867$), or soil temperature ($F = 0.00$, 1 and 2 df, $P = 0.9710$). Neither were there differences in Eh among the soil levels ($F = 0.81$, 1 and 2 df, $P = 0.4630$) or between the untreated cores ($F = 1.18$, 2

and 2 df, P = 0.4589). Soil Eh averaged -149 mV overall during prolonged, flooded storage.

CO_2 Emissions

From continuously flooded cores, CO_2-C emissions averaged 155 g m^{-2} yr^{-1} (n = 47, SE = 5) in fresh cores, 58 g m^{-2} yr^{-1} (n = 96, SE = 4) in brackish cores, and 89 g m^{-2} yr^{-1} (n = 48, SE = 5) in saline cores. CO_2 emissions were greater when water-table depths were lower, but marsh types varied in the way CO_2 emission responded to differences in water-table depth, as indicated by the marsh type by water-table depth interaction term (F = 4.75, 2 and 24 df, P = 0.0183). The rate of CO_2 emission differed least over the range of water-table depths in brackish cores, and most in fresh cores (Figure 7).

The following equations describe the relationship observed between CO_2-C emissions (y; g m^{-2} yr^{-1}) and water-table depth (x; cm):

fresh marsh: $y = 457.0 + 13.0\ x$ (1)

brackish marsh: $y = 137.4 + 5.5\ x$ (2)

salt marsh: $y = 145.5 + 15.1\ x$ (3)

These equations apply only to water-table depths at or below the soil surface. Visual inspection of the data suggested that CO_2 emissions were slightly greater during continuous drainage than during tidal like drainage at the deeper water-table depths, but no difference was indicated between static water levels and tidal like water levels (F = 0.78, 1 and 24 df, P = 0.3851).

In contrast to the untreated cores, no relationship between CO_2 emission and soil temperature was detected (F = 0.80, 1 and 24 df, P = 0.3805), which ranged only from 19.2°C to 23.4°C in treated cores.

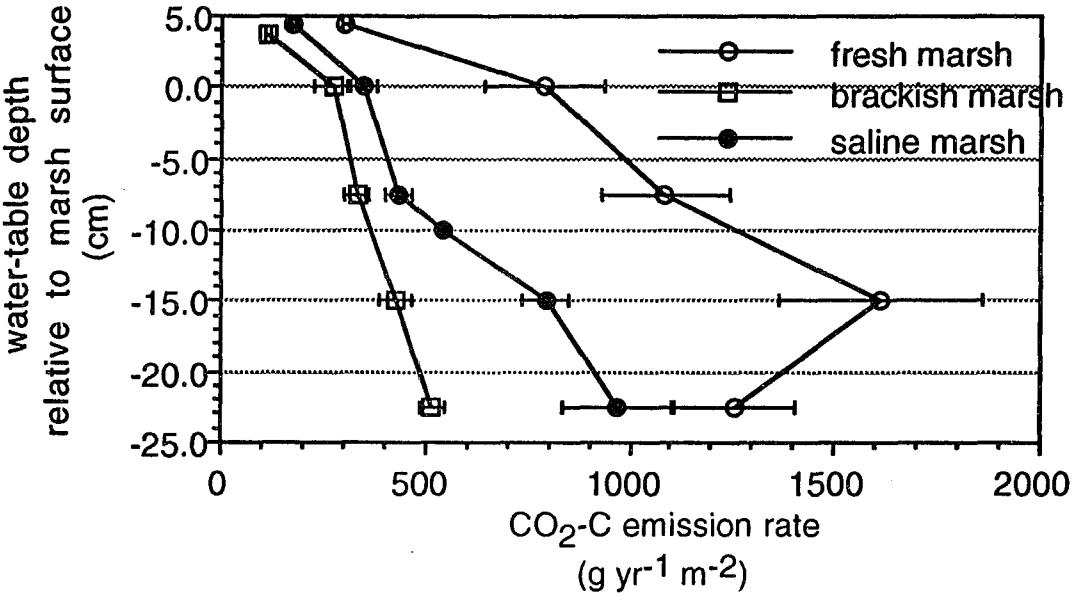

Figure 7. Relationship between water-table depth and CO_2-C emissions in fresh, brackish, and saline marsh soils, continuous and tidal-like drainage combined; each point is the mean of >12 observations, except saline marsh drained to -10 cm (n = 3).

Eh Measurements

Differences in Eh were detected among the marsh types ($F = 7.82$, 2 and 24 df, $P = 0.0024$), and post ANOVA analyses indicated that all three marsh types differed. Eh averaged over all observations differed only 24 mV between saline and fresh cores whereas Eh averaged over all observations in brackish cores was 65 mV and 90 mV higher than in fresh and saline cores respectively (Table 3). The relationship between Eh and soil level also varied among marsh types, as indicated by the marsh type by soil level interaction term ($F = 20.52$, 2 and 1584 df, $P = 0.0001$). Thus, in the upper levels of brackish cores, Eh decreased less with depth than in fresh and saline cores (Figure 8).

Eh was greater when the water-table depth was lower, but water-table depth affected Eh differently in the two water regimes, as indicated by the water-table depth by water regime interaction term ($F = 13.71$, 1 and 24 df, $P = 0.0011$); Eh was similar in the 2 water regimes when the cores were flooded, but at the deeper water-table depths, differences between the water regimes became apparent (Figures 9, 10, and 11). No relationship between Eh and soil temperature was detected ($F = 0.02$, 1 and 24 df, $P = 0.8805$).

Soil levels responded differently to water-table depth as indicated by the soil level by water-table depth interaction term ($F = 130.55$, 1 and 1,584 df, $P = 0.0001$). Thus in the upper layers of the soil, Eh at a given soil level did not respond to water-table depth unless the drainage was to that particular soil level, and in the lower levels of the soil, Eh did not respond to any water-table depth studied.

Table 3. Adjusted means of soil Eh and carbon (as CO_2) emission from fresh, brackish, and saline marsh soil cores over all treatments; C is mean of 342 observations during which water depth averaged -2.8 cm and temperature averaged 21.2 oC. Eh is mean of 1,624 measurements from soil levels ranging from -1 to -22.5 cm, average soil level was -10.2 cm.

marsh type	C (g yr-1 m-2)			Eh (mV)		
	mean	n	stderr	mean	n	stderr
fresh	780	91	22	-89	432	8
brackish	228	156	18	-24	738	6
saline	409	95	22	-114	454	8

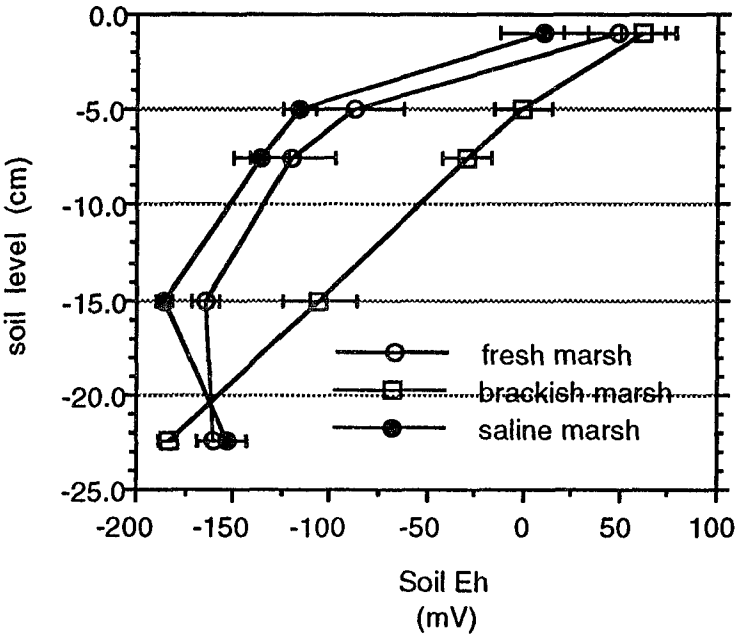

Figure 8. Adjusted mean Eh at different soil levels in fresh, brackish and saline marsh cores over all observations; flooded conditions, continuous and tidal-like drainage combined. n = 1,619, water depths ranged from -22.5 cm to 10 cm, average water depth was -2.8 cm.

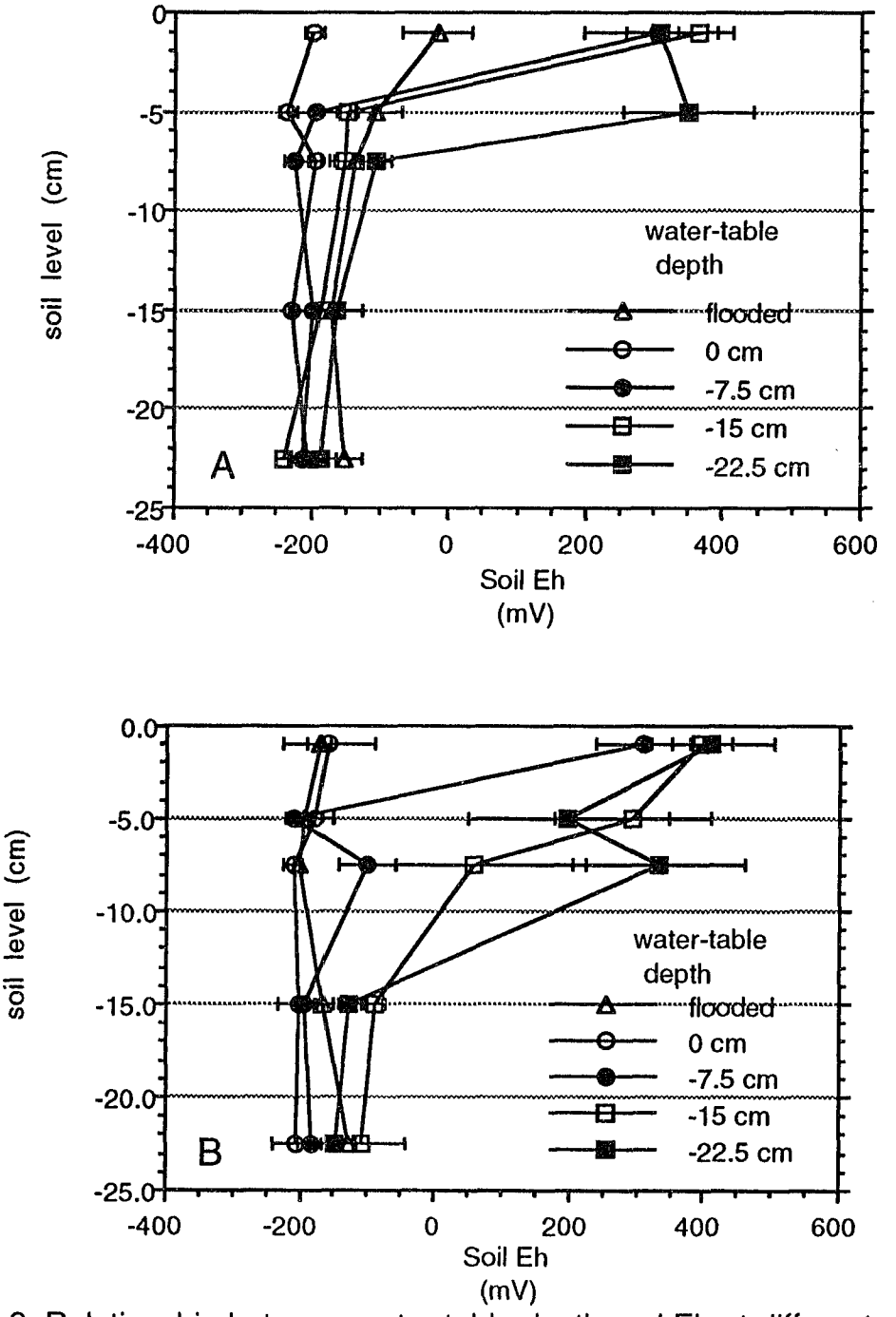

Figure 9. Relationship between water-table depth and Eh at different soil levels in fresh marsh cores with water levels held constant 72-240 h (A), and in cores alternately flooded for 12 h and drained for 12 h (B); each point is the mean of >6 observations.

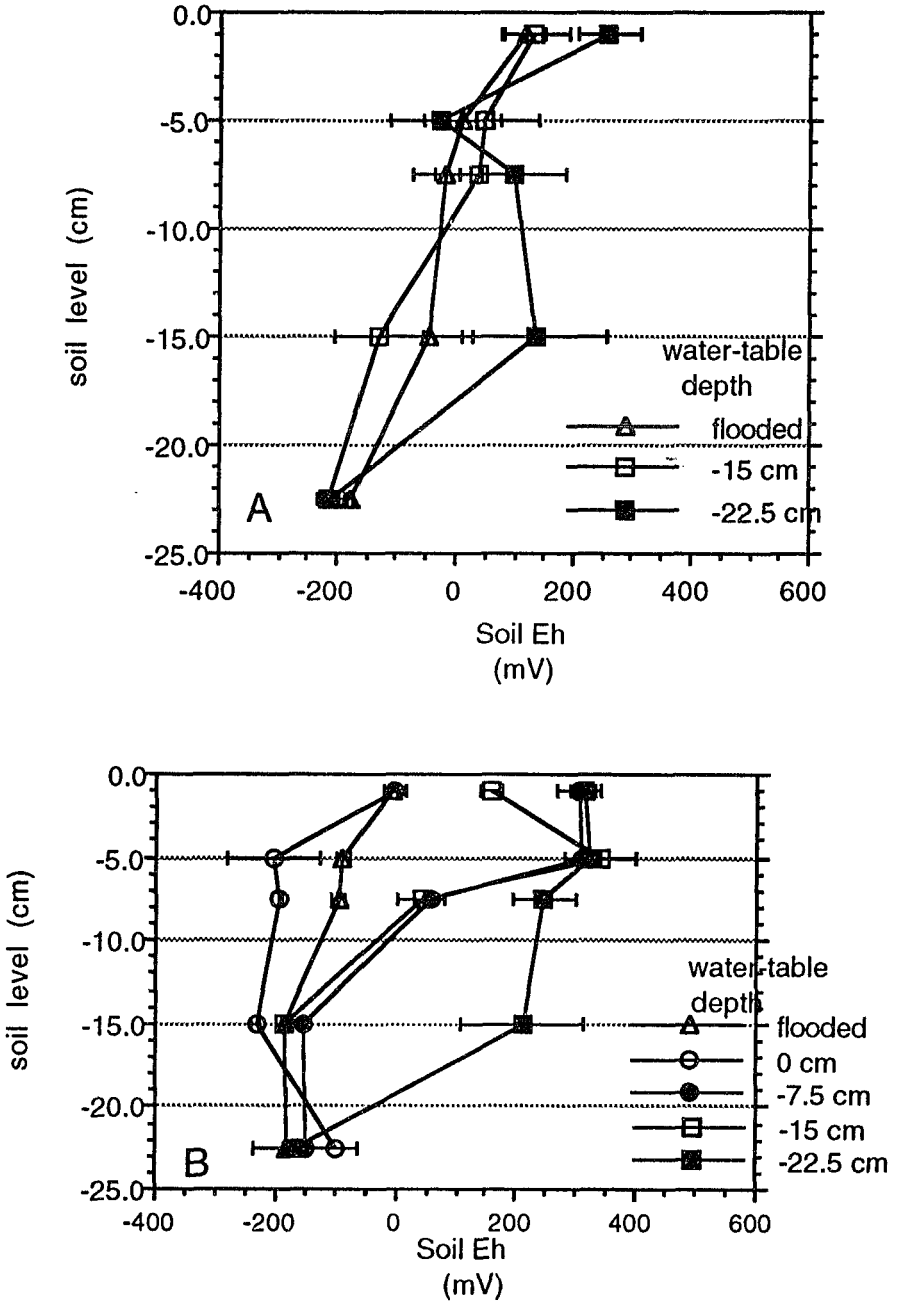

Figure 10. Relationship between water-table depth and Eh at different soil levels in brackish marsh cores with water levels held constant 72-240 h (A), and in cores alternately flooded for 12 h and drained for 12 h (B); each point is the mean of >6 observations.

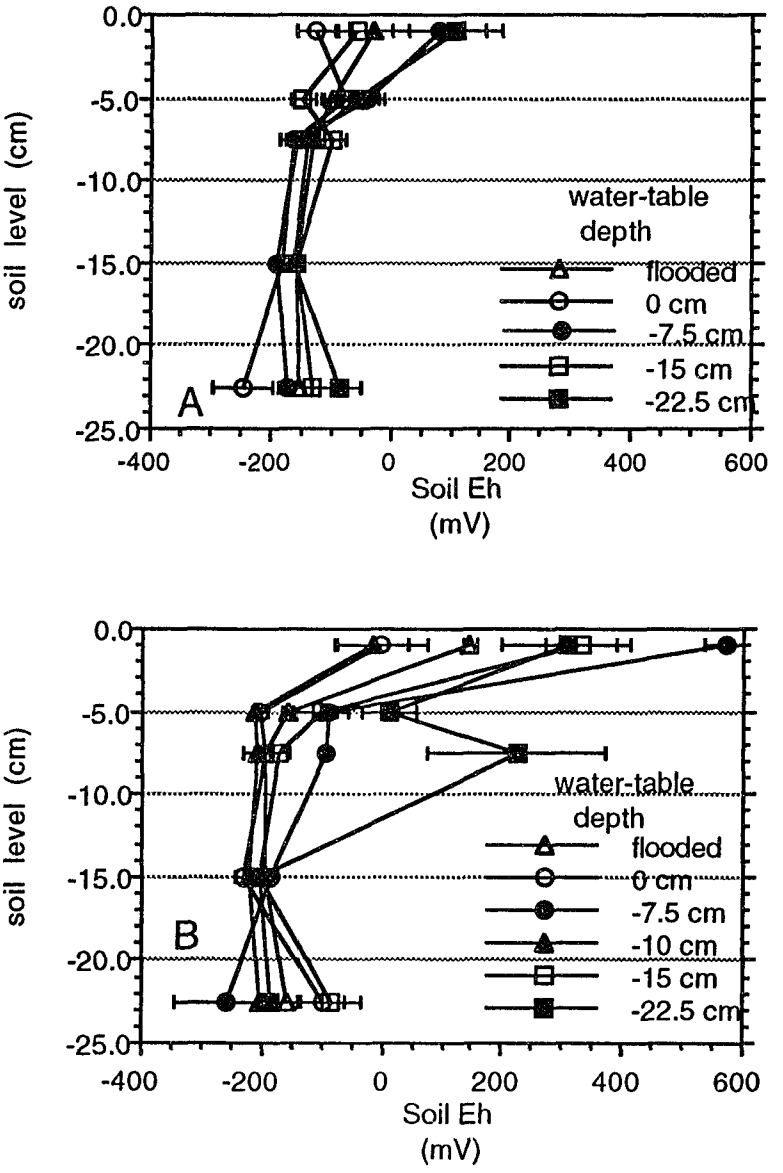

Figure 11. Relationship between water-table depth and Eh at different soil levels in saline marsh cores with water levels held constant 72-240 h (A), and in cores alternately flooded for 12 h and drained for 12 h (B); each point is the mean of >6 observation, except static water levels drained to -10 cm and to -7.5 cm (n = 3).

DISCUSSION

Untreated Cores

Failure to detect a relationship between CO_2 emissions and the length of time these cores were held in the laboratory agreed with the findings of previous workers who concluded that the soil microbial community is relatively unlinked to plant growth (Christian et al. 1978). It may at first seem surprising that depletion of soil organic matter did not occur and affect CO_2 emissions, but the organic carbon content of these soils is extremely high (Table 2). Based on reported soil bulk density and organic carbon density at sites where the cores were collected (Table 2) and the rate of CO_2 emission in these 3 flooded cores, it would take 1.4 yr to deplete the organic matter in the upper 1 cm of the fresh core, 3.5 yr to deplete the organic carbon in the upper 1 cm of the brackish core, and 3.0 yr to deplete the organic carbon in the upper 1 cm of the saline core. I am unaware of previous studies that monitored Eh in intact, flooded, highly organic soil cores over long periods of time to compare to these results, but soil Eh would not be expected to change as long as organic matter content and character were undisturbed.

Treated Cores: CO_2 Emissions

Although close agreement between estimates of CO_2 emissions derived from field studies and those determined in these cores was not necessary for the test of hypothesis, the estimates derived from these saline marsh cores were within the range of values reported for saline marsh by Howes et al. (1985), Morris and Whiting (1986), Smith et al. (1983), and Teal and Kanwisher (1961). Also, Howes et al. (1985) found no difference between CO_2 emissions from field measurements and from smaller, intact cores in the laboratory.

The relationship observed between cores from the three marsh types and CO_2 emission was similar to that of Smith et al. (1983) who found that CO_2 emission was highest in fresh marsh, and lowest in brackish marsh in the field. The significant interaction between marsh type and water-table depth ruled out the possibility that differences among marsh types were related to the different hydrological conditions among the marsh types. Soil porosity in these fresh marsh soils (96%) is greater than that in these brackish (91%) and saline (88%) marsh soils (Chapter 3), and may be partially responsible for differences in CO_2 emissions rates. The greater soil porosity in fresh marsh probably allows more contact between soil organic matter and the atmosphere, thus promoting aerobic respiration more so than in brackish and saline soils. But other factors must also have been important, otherwise there would have been a gradient of decreasing CO_2 emissions from fresh to brackish to saline cores, reflecting the gradient in porosity. Therefore, differences in CO_2 emissions were also probably related to differences in the origin of the soil organic matter. As noted, the fresh site was dominated by *Panicum hemitomon*, the brackish site by *Spartina patens*, and the saline site by *Spartina alterniflora*.

CO_2 emissions indicated that decomposition of fresh marsh plants was fastest, and decomposition of brackish marsh plants was slowest. Data from Valiela et al. (1982) also indicated that *Spartina patens* decomposed much slower than either tall or short *Spartina alterniflora*, and Odum (1988) concluded that fresh marsh plants decompose much faster than saline marsh plants. Recently, Kelly et al. (1990) reported that carbon remineralization, as measured by CO_2 and CH_4 emissions, was greater at freshwater sites than

at saline sites, and attributed the differences to the different sources of organic matter along the gradient.

It was not surprising that the rate of CO_2 emission was greater when the water-table depth was deeper because the greater degree of drainage would have increased the exposure of soil organic matter to O_2, and promoted aerobic respiration. The results of this laboratory study indicate that decreasing the water-table depth from the surface to -15 cm would increase CO_2-C emissions 43% in fresh marsh, 60% in brackish marsh, and 156% in saline marshes. Although environmental conditions may interact in the real world in ways that cannot be anticipated based on this laboratory study, this may be important because organic C accumulation was reported to be less in saline marsh soil than in fresh marsh soil under field conditions (Table 1). However in absolute terms, CO_2-C emissions would still be 75% greater in fresh marsh than in saline marsh; thus from the perspective of the global carbon balance, water-table depth may be more important in fresh marsh soil. It should also be noted that failure to detect a difference between the two water regimes indicated that the microbial response to soil drainage was complete in a short period of time.

Treated Cores: Eh Measurements

Eh increases in response to drainage have also been reported by Howes et al. (1986). The difference in Eh between the tidal-like and static water regimes, and lack of difference in CO_2 emissions between the regimes, indicated that Eh continued to respond to soil drainage after changes in CO_2 emissions were complete. This suggested that these changes originated from the chemical oxidation of reduced substances as the soil began to dry, rather than from a lessening of the biological demand

for electron acceptors. This effect was further investigated. The Eh data from the upper levels (surface, -5.0 cm, and -7.5 cm) of drained cores were analyzed to test for changes in Eh in relation to the length of time since drainage. No relationship was found between time since drainage and Eh within either tidal-like (4 to 12 h after drainage) or static water regimes (72 to 240 h after drainage) (tidal-like: $\underline{F}$ = 1.38, 1 and 9 df, $\underline{P}$ = 0.2695, static: $\underline{F}$ = 0.56, 1 and 8 df, $\underline{P}$ = 0.4775). Either variability was too great within the treatments or differences in Eh between tidal-like and static water regimes originated between 12 and 72 h after drainage. Howes et al. (1986) also noted that Eh continued to increase after drainage and suggested that this resulted from increased O_2 diffusion through newly created gas channels. If marsh soils in the field could similarly drain, this process could be important in oxidizing reduced compounds that may accumulate in marsh soils throughout the growing season (Feijtel et al. 1988). This process might only be important in the streamside zone however, because little soil drainage occurs more than 4 m from the streamside (Agosta 1985).

The failure for Eh measurements below the -7.5 cm level to respond to any degree of drainage used in this study may have resulted from the moist soil conditions that persisted throughout the treatments, probably maintained via capillary action. In very fine-grained soils, capillary action can saturate soil above the water table (Fetter 1988:91), and as noted, soil in the marsh interior often remains saturated to the near surface in spite of tidal drainage of creeks (Agosta 1985).

Eh in the upper levels of brackish cores was 90-100 mV higher than in fresh and saline cores under the same hydrological conditions. This suggests that with similar water-table depths, vegetation would be exposed

to less waterlogging stress in brackish marsh than in fresh and saline marsh. However, waterlogging stress also results in part from the presence of S^{2-}, and the effects low Eh alone are not as severe as the effect of low Eh and sulfides on photosynthesis (Pezeshki et al. 1988) and growth (Pearson and Havill 1988). The rate of reduction of SO_4^2 to S^{2-} is partially dependent on the availability of SO_4^{2-} (Nedwell 1982), and the SO_4^{2-} concentration in these marshes increases from fresh to brackish to saline marshes (Brupbacher et al. 1973). Thus, even though Eh may be expected to be similar in fresh and saline marshes, saline marsh vegetation would likely be exposed to more severe waterlogging stresses.

It was not clear why Eh was not the same in all marsh types. Some of the difference may have resulted from differences in porosity among the marsh types. As mentioned, soil porosity decreases from these fresh to brackish to saline marshes (Chapter 3), which probably allowed more contact between fresh marsh soil and the atmosphere. But other factors must also have been important, otherwise there would have been a gradient of decreasing Eh from fresh to brackish to saline cores. Much of the differences must therefore result from differences in decomposition rates, and from differences in behavior of different reduced electron acceptors. The lower rate of decomposition in brackish marsh soil decreased the demand for terminal electron acceptors, which probably caused higher Eh than in the other two marsh types. CO_2 emissions were much greater in fresh marsh than in saline marsh, but Eh was slightly lower in saline marsh. Odum (1988) concluded that Eh was lower in saline marsh than in fresh marsh, and attributed the lower Eh in saline marsh to the presence of the sulfur redox couple.

CONCLUSIONS

This study tested the hypothesis that hydrological conditions were responsible for reported differences in CO_2 emissions among fresh, brackish and saline marsh soils. Although hydrological conditions greatly modified CO_2 emissions, inherent differences in CO_2 emissions as well as soil Eh were found among the marsh types. It remains to be seen if this conclusion applies beyond the individual marshes sampled. A number of variables were ruled out as the cause, and differences in CO_2 emission and soil Eh were attributed to differences in the character of the organic matter that is decomposing in fresh, brackish, and saline marsh. However, studies specifically designed to test that explanation are needed before conclusions are made.

SECTION III: FIELD STUDIES AT THE LAKE BARRE MARSHES

CHAPTER 5
RATIONAL FOR FIELD WORK

Field studies were conducted in a rapidly deteriorating marsh to determine if the soil formation requirements and relationships among mineral matter, decomposition, soil Eh, and marsh type identified in Chapter 3 and 4 were related to marsh deterioration. Perhaps the deteriorating marshes were failing to vertically accrete. If so, why? As noted in Chapter 1, marsh vertical accretion was perceived to operate as negative feedback loop that maintained the marsh surface elevation within a few centimeters of water levels. Perhaps mineral matter accumulation was inadequate, or too much organic matter was decomposing or too little was being produced. Perhaps vertical accretion was adequate, but the vegetation was dying because of anoxic soil conditions or salt stress. Perhaps there was too little soil mineral matter for vigorous plant growth, or too little soil mineral matter in brackish areas experiencing saltwater intrusion. Understanding the mechanism of marsh loss required detailed study of vegetation and soil. This Section contains two separate studies. One examined organic matter cycling in the marsh plant community because of the importance of above ground production, belowground production, and decomposition. The other study examined marsh soil formation in the study area. That study determined if vertical accretion, mineral matter accumulation, and organic matter accumulation were adequate, and determined what controlled vertical accretion.

STUDY AREA

The study area was in southeast Louisiana, between Bayou Terrebonne and Bayou Barre (Figure 12). To the south progressively lie Lake Barre, Terrebonne Bay, and the Gulf of Mexico. The study area is part of the Lafourche delta complex. Bayous Terrebonne and Barre are distributaries of Bayou Lafourche, which shared the flow of the Mississippi River from roughly 4,200 years ago until 1904 when it was dammed at its upstream end (Frazier 1967). The most recent period of marsh building started roughly 800 years ago when the Mississippi River reoccupied Bayou Terrebonne, which prograded into shallow bays (Frazier 1967). The area is now in the delta lobe abandonment phase (Coleman and Gagliano 1964, Morgan 1967, Coleman 1988) and sediment dewatering and compaction cause subsidence as marine processes replace riverine process. This is the reason that this area was used. During abandonment, sediment starvation and continued compaction of the underlying sediments leads to submergence and transgression, i.e., salt-water intrusion (Coleman 1988, Morgan 1967). Pipeline canals indicate that petroleum extraction has also occurred.

Marsh loss rates in the study basin continue to increase even though marsh loss rates recently decreased throughout much of Louisiana (Britsch and Kemp 1990). Conversion of marsh to open water averaged 320 ha per year between 1974 and 1983 for the entire USGS Dulac 15 minute series topographic map (Britsch and Kemp 1990). Much of this loss was concentrated in a large hotspot (roughly 600 ha) near midbasin where broken marsh was indicated on the 1964 USGS 15 minute series topographic map (Dulac quadrangle). The wetland loss maps prepared by May and Britsch

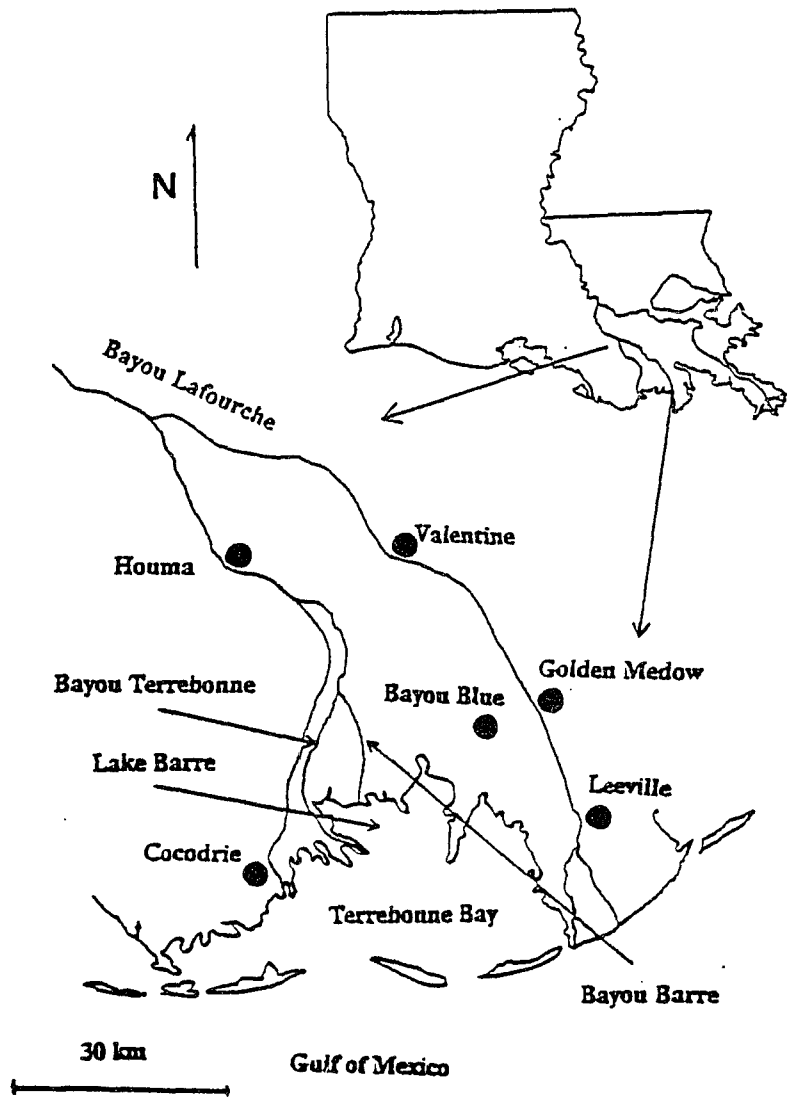

Figure 12. Location of Lake Barre area and tide gauge stations used to estimate submergence.

(1987) indicated that this hotspot formed after 1974. The landscape pattern was not a factor in selection of the study area.

Rapid subsidence causes rapid marsh submergence in southeast Louisiana (Penland and Ramsey 1990). Our study sites experienced rapid submergence, as indicated by tide gauge analyses. Tide gauge analyses indicated that submergence averaged 1.38 cm/yr since 1964 at 6 tide gauges surrounding the basin that contained our study sites (Table 4). This time period was used because it is the same time span over which vertical accretion was estimated. The most likely cause of the rapid submergence is the thick deposit of under-consolidated Holocene sediments in the study area, which are 180 m thick (Penland et al. 1988). Petroleum withdrawal has also occurred in the study area, but whether it increased subsidence in the study area is unknown.

The submergence rate of interior marshes may be even greater than indicated by comparing our marsh surface vertical accretion rates to tide gauge submergence rates.. This is because unlike tide gauge sites, the marsh surface is affected by oxidation of surface peats (DeLaune et al. 1990) and autocompaction of underlying peats (Kaye and Barghoorn 1964). The tide gauges are anchored deeper and in more stable levee sediments adjacent to major Mississippi River distributaries that are more consolidated than bay fill and marsh deposits. Surface settling increases as peat thickness increases, and data available to Kaye and Barghoorn (1964) suggested that peat thicker than 9 m caused autocompaction rates too great for marsh accretion to counter. Peat in this study area ranges only up to 1.5 m, but lies on an additional 180 m of unconsolidated alternating Holocene bay fills and peats (personal communication, S. Penland, Louisiana Geological Survey).

Table 4. Submergence rates in the Lake Barre/Madison Bay area of Terrebonne Parish Louisiana, estimated from tide gauge analysis, from Penland et al. (1988).

Station (cm/yr)	Period of Record	Submergence Rate
Houma, Intracoastal Waterway	1962-1982	1.94 ± 0.39
Cocodrie, Bayou Petit Caillou	1969-1983	0.60 ± 0.32
Valentine, Bayou Lafourche	1966-1982	1.62 ± 0.29
Bayou Blue, near Catfish Lake	1976-1983	1.02 ± 0.46
Golden Meadow, Bayou Lafourche	1959-1979	2.33 ± 0.27
Leeville, Bayou Lafourche	1957-1983	0.74 ± 0.12
---mean of these stations---	1964-1982	1.38 (s. d. = 0.70)

Salt-water intrusion is also common in southeast Louisiana. Chabreck (1970) compared a vegetative map made in the 1940's to one made in 1968 and found that the saline marsh type had moved inland an average of 3.4 km during the interim. Much of this study basin was located in areas that were brackish marsh in the 1940's (O'Neil 1949) and whose soil was classified as brackish marsh, deep peat in the 1950's (USDA 1956). However, the border between saline and brackish marsh has been migrating northward and westward such that it has gradually moved 6-7 km between 1940's and the 1980's (O'Neil 1949, Chabreck and Linscombe 1978, Chabreck and Linscombe 1988).

I noted during preliminary visits that soil was too weak to walk on without sinking knee or thigh deep even in saline marsh. In *Spartina patens* marsh, rooted plant stems were clumped rather than uniformly or randomly distributed. Plants occurred only in hummocks, which are small areas that are slightly more elevated than the unvegetated areas. Hummocks were generally <1.0 m^2 in area and were separated by 0.5 to 5 m. Hummocks were smaller, had vertical sides and were closer to each other at the more brackish areas, but were larger, had more sloping sides and were farther apart in the more saline areas. I estimated that 7/8 of the marsh surface had no rooted stems. Decumbent *S. patens* stems covered much of the area lacking rooted stems in the northern end of the basin, but in the midbasin at the southern extent of the *S. patens* marsh. The hummocks appeared to be built entirely of densely packed, living roots. Smaller hummocks were light enough to be picked up and moved with no apparent damage to living roots. Much sampling was done from a pirogue (a flat bottomed, canoe like boat).

CHAPTER 6

ORGANIC MATTER CYCLING AND MARSH STABILITY IN A RAPIDLY SUBMERGING ESTUARINE MARSH

INTRODUCTION

Marsh loss is generally associated with plant stress. Furthermore, plant production is needed for soil formation (Chapter 3), healthy plants are generally believed necessary to hold these weak soils in place, and marsh plant production can also be important to estuarine fish at some stages in the life cycle (Deegan et al. 1990). Thus, the purpose of this study was to estimate organic matter pools and fluxes among major components of the carbon cycle in tidal, non-saline marsh experiencing rapid submergence and marine intrusion.

Two sites were selected that were separated by approximately 3 km. Site selection was constrained by the opposing goals of being near each other to minimize differences between them in subsidence rates, but far from each other to maximize differences between them in marine intrusion. The sites will be referred to as the less saline site and the more saline site, with the more saline site being more southerly and more closely linked to the Gulf of Mexico (Figure 13). The hotspot was approximately 2 km SW of the southern site. I assumed that the two sites experienced equal submergence because of their proximity, but there was no way to test that assumption.

Both sites were located in areas that were brackish marsh in the 1940's (O'Neil 1949) and whose soil was classified as brackish marsh, deep peat in the 1950's (USDA 1956). However, the border between saline and brackish marsh has been migrating northward such that it was 2-3 km south of the

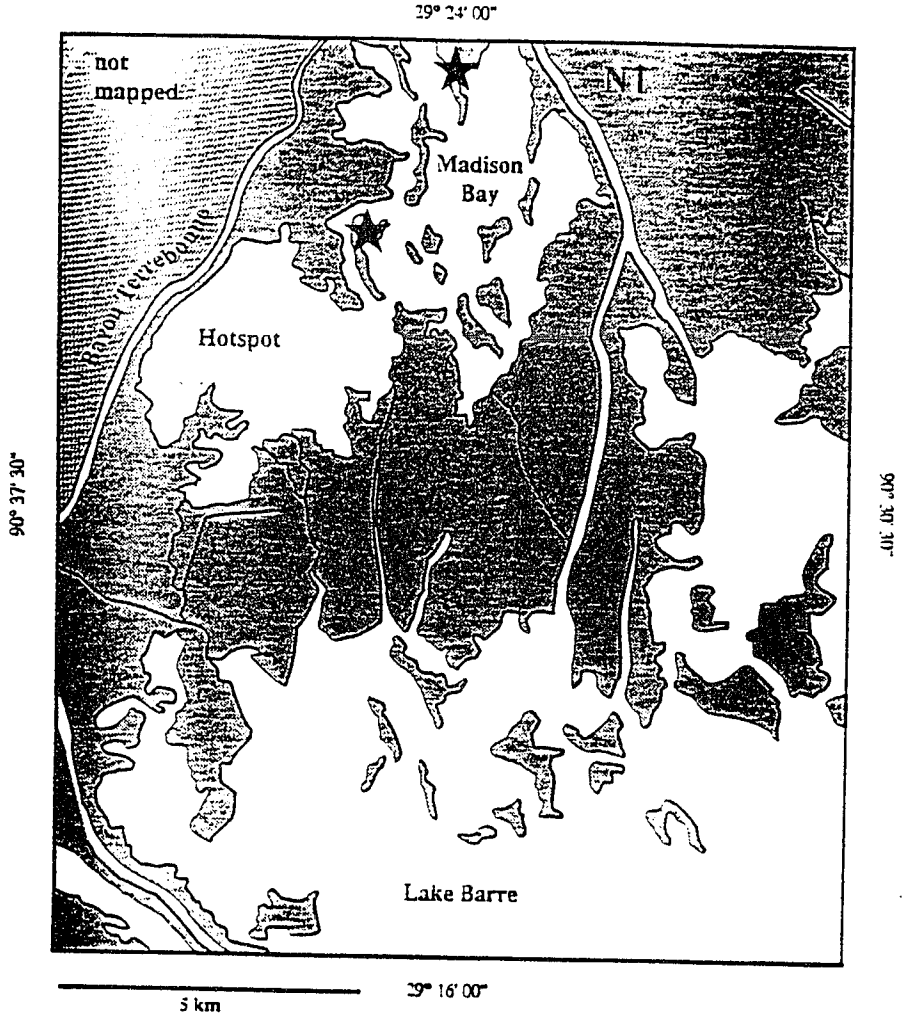

Figure 13. Location of study sites where organic matter cycling was studied in the Lake Barre study area.

more saline site in the 1940's (O'Neil 1949), at the more saline site in 1978 (Chabreck and Linscombe 1978), and at the less saline site in 1988 (Chabreck and Linscombe 1988). Thus, salt-water intrusion occurred at both sites, but was more advanced at the more saline site.

METHODS

Above-ground Production

Biomass was harvested for 2-years. Litter and all rooted stems were harvested spring (March), summer (June/July), and fall (September) during 1990. Seven harvests were made in 1991: March, April, May, June, July, August, and October. Four plots were harvested from each marsh during each visit. Previous researchers in Louisiana brackish marshes harvested from 0.25-m^2 plots (White et al. 1978) and 0.1-m^2 plots (Hopkinson et al. 1978). I used larger plots to prevent samples from lacking vegetation. In 1990, 1-m^2 plots were harvested, but in 1991 plot size was reduced to 0.5 m^2 to reduce processing time of samples. Samples were returned to the lab, refrigerated for storage, and classified as live or dead culms by species. Samples were then oven dried. Median live biomass by species and total live and dead biomass were compared between marshes with Wilcoxon's signed ranks test (Steele and Torrie 1980:533-540). This test for paired observations is non-parametric and therefore does not require that data are normally distributed. Annual above-ground production over the 2 year period was estimated from the oven-dried weight of live and dead material with Smalley's method (Shrew et al. 1981).

Below-ground Production

Below-ground biomass was sampled at the same times that above-ground biomass was sampled. Cores (15-cm diameter, 20 cm long) were collected from each harvested plot. Previous estimates of below-ground biomass typically used cores ranging from 6.0 to 10-cm diameter (de la Cruz and Hackney 1977, Jordan et al. 1989). Cores were collected from hummocks (vegetated) and unvegetated marsh soil during 1990. Core collection from unvegetated marsh was not continued during 1991 because live biomass was rarely encountered in those samples. During 1990, samples were washed with water and sodium metaphosphate to remove soil; live and dead material were separated according to Hopkinson and Dunn (1984). Live and dead biomass were clearly distinguishable in some samples, but in most samples there appeared to be a gradient from live to dead, with much biomass appearing somewhere in between. Other workers have also reported difficulty estimating live belowground biomass and instead reported belowground biomass as a combination of live and dead material (see Good et al. 1982). Live and dead seperation efforts were therefore disontinued after 1990. Instead, cores were dried, weighed to determine bulk density, ground, and samples were combusted to estimate percent organic matter (Davies 1974). Total organic matter content of the cores was then estimated from soil bulk density and percent organic matter. Median below-ground biomass was compared between marshes with Wilcoxon's signed ranks test (Steele and Torrie 1980:539-540). Blowground biomass were analyzed for differences between vegetated and unvegetated marsh with Wilcoxon's signed ranks test (Steele and Torrie 1980:539-540). Annual below-ground production over the two year period was estimated from changes in total below-ground biomass in

a manner similar to Smalley's method. If the change in total biomass was positive, then production for that interval equaled that change; if the change was zero or negative, then production was zero.

Respiration

CO2 emission rates were determined from vegetated and unvegetated marsh soil at both sites in April 1990 and September 1990 with a flow through Infrared Gas Analyzer as described in Chapter 4, with the exception that these measurements were made from the marsh surface rather than from intact cores in the lab. These field measurements were made over litter as well as soil. The CO_2 emission rate measured in the field was corrected for seasonal differences and for annual CH_4 emission as noted by Smith et al. (1983) to estimate total C emissions. A factor of 1.724 was used to estimate organic matter from carbon (Wilson and Staker 1932).

Burial

Two cores (15 cm diameter, 50 cm long) were collected from hummocks in each marsh. These cores were part of 30 collected to study sedimentation throughout this area (Chapter 7). Cores were returned the lab and sectioned into 3-cm increments. Soil bulk density was determined from the oven dried weight of soil increments. Total vertical accretion on the marsh surface since 1963 was estimated from the depth of the soil increment containing the ^{137}Cs maxima, and the number of years between core collection and 1963 (DeLaune et al. 1978). Vertical accretion averaged over greater depths underestimates total vertical accretion (changes relative to shallow horizons) because of autocompaction, which causes marsh surfaces to settle (Kaye and Barghoorn 1964). Vertical accretion averaged over greater depths is required to estimate net vertical accretion (changes in absolute

elevation), which was not estimated in this study. ^{137}Cs activity was estimated from the gamma ray emission rate (661.7 KeV) from soil increments; standard deviations were also determined (Wang et al. 1975:300). Entire soil increments were counted in a Marinelli beaker that fit over a 7.6 cm by 7.6 cm Li drifted Ge detector crystal. The ^{137}Cs maxima were assumed to result from atmospheric deposition of ^{137}Cs that mark the 1963 surfaces, rather than from deposition of eroded, ^{137}Cs containing sediments that mark 1964 surfaces, as occurs in some reservoirs (Ritchie et al. 1973). This is because ^{137}Cs profiles in Louisiana marshes represent continual atmospheric deposition rather than discontinuous sediment deposition (personal observation). If I erred, then vertical accretion was underestimated 4%.

^{137}Cs mobility is not generally a problem because illites, the most common clays in nature (Millot 1970:8), bind ^{137}Cs internally so that only destruction of the clay lattice (e.g., digestion with 5 N nitric acid) causes desorption (Lomenick and Tamura 1965). Lomenick and Tamura (1965) found 84% of the ^{137}Cs in their soil bound to illites even though illites accounted for only 35% of their sediments. Illites are the major clays in the Atlantic Ocean and make up 40-50% of the clays in the Gulf of Mexico (Griffin et al. 1968). Illites are common off the Louisiana coast (Brooks et al. 1976), in Louisiana estuarine sediments (Gambrell et al. 1977), and account for 10-30% of mineral matter in Louisiana marsh soils (pers. comm. Dr. W. Hudnall, Department of Agronomy, Louisiana State University Agricultural Center, Baton Rouge, Louisiana). X-ray diffraction analyses of soil from the northern extreme, southern extreme, and middle of this study area indicated that illites were present in all samples (pers. comm. Dr. Alan Bailey, Department of Geology, University of Southwestern Louisiana, Lafayette, Louisiana). Illites are not

common in freshwater bodies of the southeastern United States (Neiheisel and Weaver 1967), and Alberts et al. (1979) noted ^{137}Cs mobility in a South Carolina freshwater reservoir that was subsequently explained by a lack of illites (Evans et al. 1983). See Ritchie and McHenry (1990) for a recent review of ^{137}Cs dating applications.

Soil increments that contained the 1963 marsh surface, and all overlying increments were ground. Samples from each increment were combusted to determine percent organic matter (Davies 1974). The amount of organic matter in each soil increment was then estimated from the soil bulk density and percent organic matter of each increment. Annual burial rates were then estimated from the total amount of organic matter buried since 1963, and the number of years between core collection and 1963 (27).

Export

Export was estimated from the difference between production and the sum of decomposition plus burial. Other studies have also estimated export indirectly, e.g., Day et al. (1973) estimated export the difference between production and respiration.

Edaphic Conditions

Water salinity was measured either in the adjacent bayou or on the marsh surface during most site visits with a YSI salinometer. Differences in salinity between bayou water and water standing on the marsh surface were negligible. Soil water salinity was not measured. Salinity was also recorded at these sites when sampling for other studies in this area, and often there were more than one salinity measurement for each month that were averaged to obtain monthly estimates. Water salinity was compared between sites with Wilcoxon's signed rank test, pairing on month (n = 8). Soil Eh was recorded

during spring and fall site visits 1990 with duplicate platinum electrodes as described by Faulkner et al. (1989) in vegetated and bare soil at -1 cm, -5 cm, and -10 depth. Soil Eh was compared between marshes with Wilcoxon's signed ranks test, pairing on depth and sampling date (n = 6). Depth of water standing on the marsh surface was also recorded. These data were used to estimate average flooding depth as well as hummock elevation relative to the unvegetated marsh surface.

RESULTS

Hummocks appeared bigger at the less saline site and averaged 16 cm above bare soil; hummocks averaged only 12 cm above bare soil at the more saline site. *S. patens* greatly dominated the less saline site, but not the more saline site (Figure 14). Total live standing crop was greater at the less saline site than at the more saline site ($P > T^+ = 0.0137$) as was live *S. patens* ($P > T^+ = 0.0010$) and total dead biomass ($P > T^+ = 0.0332$). However, the more saline site had more *S. alterniflora* biomass ($P > T^+ = 0.0244$) than the less saline site (Table 5). No difference was detected between the sites in biomass of *Distichlis spicata* (L.) Greene ($P > T^+ = 0.1162$), and no other species were harvested. Aboveground production was greater at the less saline site than at the more saline site (Table 6).

Belowground biomass was also greater ($P > T^+ = 0.0420$) at the less saline site than at the more saline site (Table 5). Belowground biomass was not significantly correlated to aboveground biomass within the range sampled (unpublished data), and there was 7.5 and 10.4 time more belowground biomass than aboveground biomass at the less saline and more saline sites respectively. There was roughly twice as much belowground biomass in

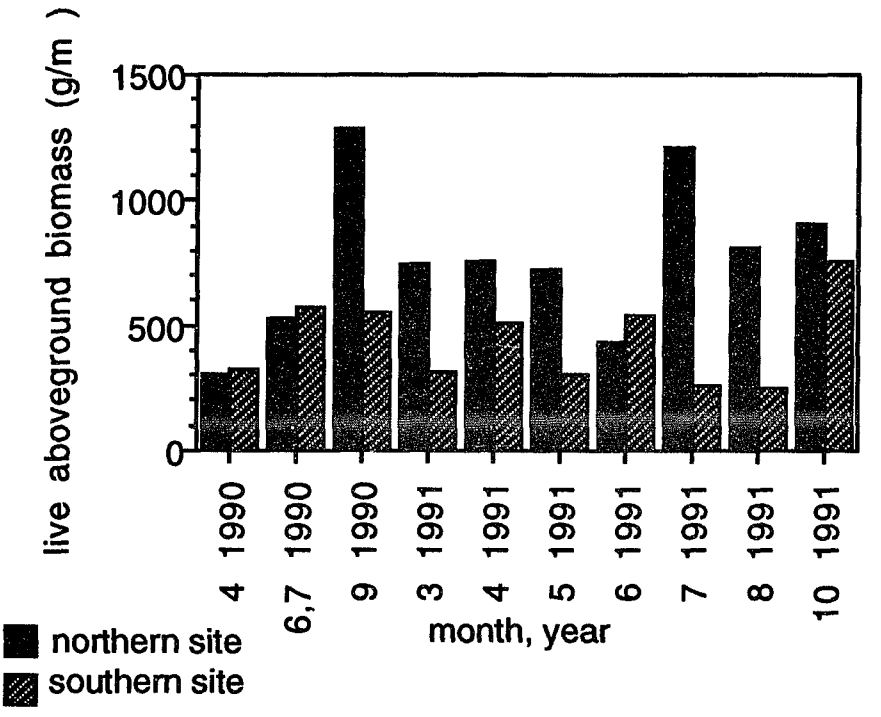

Figure 14. Live biomass of *S. patens* and *S. alterniflora* on sampling dates in 1990 and 1991 at two tidal, non-saline marshes in southeast Louisiana. Each observation is the mean of 4 plots.

Table 5. Vegetation biomass (g m^{-2}) at 2 marsh sites north of Lake Barre, Louisiana; 4 plots per site per sampling date , 10 sampling dates between March 1990 and October 1991.

Species	mean	median	range
North Madison Bay marsh			
live *Spartina patens*	604.0	598.1	226.4 - 1,063.0
live *Spartina alterniflora*	45.8	24.6	2.3 - 167.2
live *Distichlis spicata*	101.2	89.4	25.4 - 275.2
total live biomass	769.4	747.2	304.0 - 1,287.7
total dead biomass	641.5	569.5	266.0 - 1,351.9
belowground biomass	10,628.9	9,924.0	8,140 -14,788
West Madison Bay marsh			
live *Spartina patens*	207.9	209.0	88.8 - 324.4
live *Spartina alterniflora*	163.9	110.9	36.9 - 595.6
live *Distichlis spicata*	63.14	50.4	17.8 - 153.5
total live biomass	436.6	417.6	247.0 - 755.4
total dead biomass	508.3	333.0	171.4 - 1,217.6
belowground biomass	9,809.9	9,487.0	7,473 - 12,030

Table 6. Organic matter cycling (g m^{-2} yr^{-1}) estimated from sequential biomass harvesting at 2 marsh sites north of Lake Barre, Louisiana, 1990-1991.

	less saline	more saline
above ground production	1,252	681
belowground production	1,401	585
total production	2,653	1,266
soil respiration	483	734
burial	796	434
export	1,377	98

hummocks as in areas lacking rooted stems ($P > T^+ = 0.0156$). Annual belowground production was 281 g m^{-2} and 0 g m^{-2} in unvegetated marsh at the less saline and more saline sites respectively. Annual belowground production in hummocks was extremely high; 8,960 g m^{-2} and 4,678 g m-2 at the less saline and more saline sites respectively. Annual belowground production reflects the relative contributions of hummocks and bare marsh to marsh surface area (Table 6).

Some of the differences in plant community structure and function may be related to differences in flooding and salinity between the sites. Soil was more reduced at the more saline site than at the less saline site ($P > T^+ = 0.0156$), suggesting a greater frequency of flooding at the more saline site (Figure 15). Averaged over all 3 depths, soil Eh averaged -17 $\pm$ 42 mV at the less saline site and -126 $\pm$ 15 mV at the more saline site. Recorded water depths at the less saline site ranged from 10 cm to 53 cm over bare soil, and -5 cm to 17 cm over hummocks. Water salinity also appeared to be greater at the more saline site. When salinity estimates differed between the sites by more than 0.5 ppt (5 of 8 months) salinity was always greater at the more saline site (Figure 16). However, salinity at the more saline site (6.2 ppt) was not significantly different ($P > T^+ = 0.0977$) from that at the less saline site (5.2 ppt). Salinity was observed to fluctuate greatly, particularly at the more saline site. For instance, salinity at the more saline site increased from 6.2 ppt to 12.5 ppt in just 7 days during April 1990. Such fluctuations may be more of a factor on plant growth than mean salinity, but too few data were collected to evaluate that effect. Salinity also appeared lower in 1991 than in 1990 (Figure 16), but that effect was not evaluated.

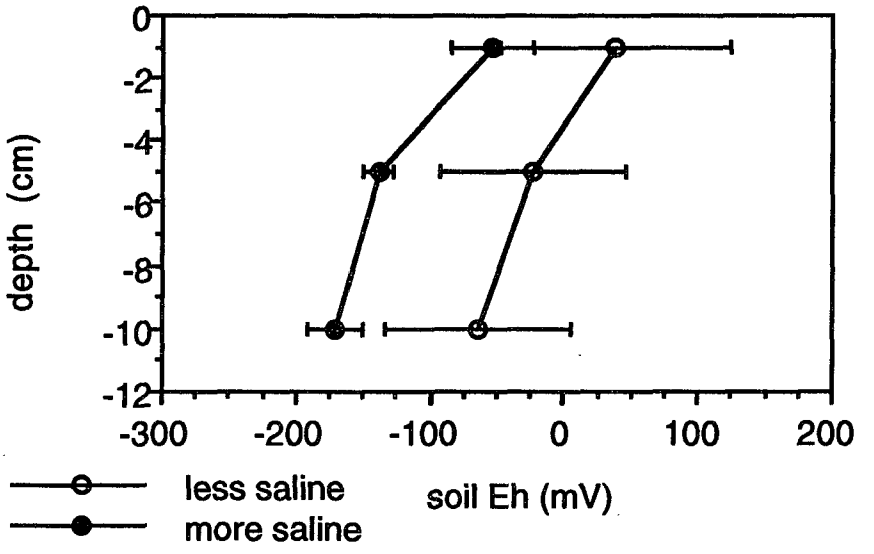

Figure 15. Soil Eh with depth at 2 marsh sites north of Lake Barre, Louisiana, 1990-1991.

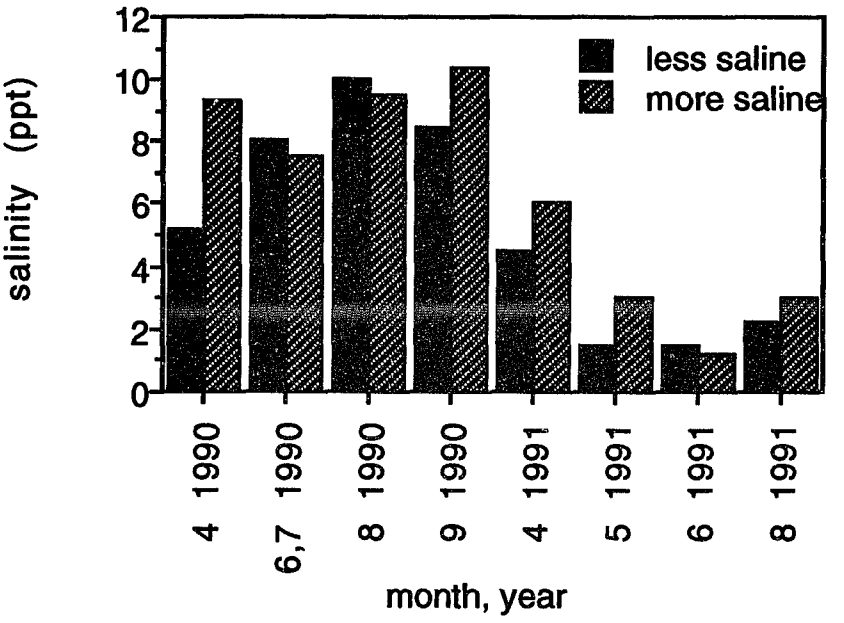

Figure 16. Salinity observed at 2 marsh sites north of Lake Barre, Louisiana, 1990-1991.

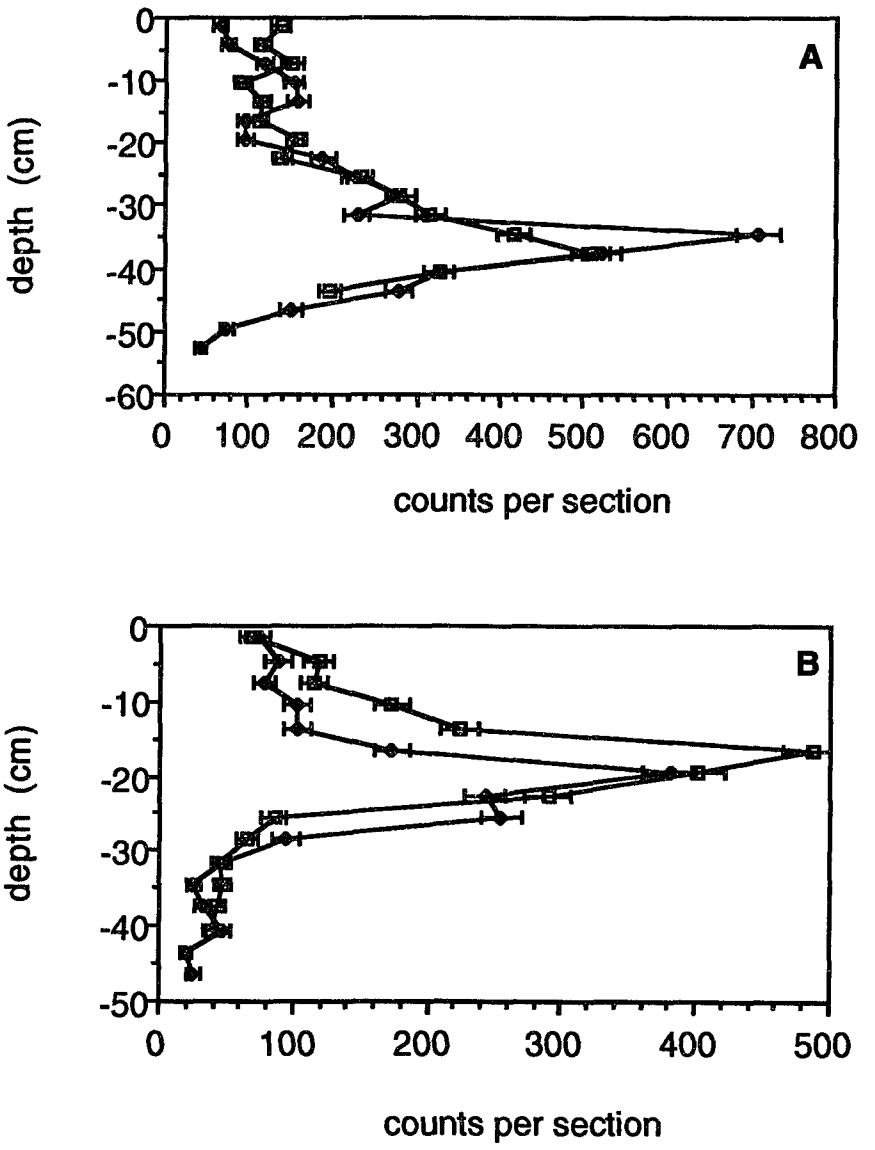

Figure 17. ^{137}Cs activity with depth in replicate cores from2 marsh sites north of Lake Barre, Louisiana, 1990-1991. Peak ^{137}Cs activity marks the 1963 marsh surface. A represents the less saline site, B represents the more saline site

Respiration was greater at the more saline site than the less saline site, in spite of more reducing conditions at the more saline site (Table 6). Respiration was generally greater over hummocks than over bare marsh soil, but too few samples were collected to statistically evaluate that effect. Organic matter burial was greater at the less saline site, where production was greater and decomposition was lower. Vertical accretion was greater at the less saline site than at the more saline site (Figure 17), and is the most likely reason for the flooding differences between sites. Organic matter available for export was greater at the less saline site than at the more saline site. Given the uncertainty inherent in summing these estimates, it is possible that the more saline site was actually neutral.

DISCUSSION

S. patens biomass was generally lower than reported elsewhere in Gulf Coast brackish marshes (Hopkinson et al. 1978, White et al. 1978), especially at the more saline site. S. *patens* biomass did not change in a regular manner, as others have also noted in Gulf Coast brackish marshes (Hopkinson et al. 1978). *S. alterniflora* biomass was also lower than previously reported in Gulf Coast saline marshes, even at the more saline site where it was more common. Low plant biomass is commonly related to flooding and associated stresses (Chalmers 1982). As noted, these sites appeared to be almost continually flooded.

Root-shoot ratios at both sites were greater than previously reported for monospecific S. *patens* (Knox 1986a:51). High root-shoot ratios are indicative of unfavorable soil conditions (Knox 1986a:49) and are further evidence that these marshes flooded excessively. Marsh plants appear to put

more energy into root production under unfavorable soil conditions (Mitsch and Gosselink 1984:195), perhaps because unfavorable soil conditions require greater root surface to service each unit of aboveground biomass (Good et al. 1982).

Excessive flooding was also indicated by the low soil Eh observed. DeLaune et al. (1983) observed soil Eh in the upper 10 cm of soil in a short *S. alterniflora* marsh approximately 100 mV more oxidized that I observed even at the less saline site where Eh was greater. The difference in soil Eh between the less saline and more saline site indicated that the more saline site was more flooded than the less saline site. It should be noted that the lower soil Eh at the southern site may also partly result from the greater amount of *S. alterniflora* at that site. The study documented in Chapter 4 showed that Eh could be lower in *S. alterniflora* marsh soil than in *S. patens* marsh soil even when soil drainage was the same, likely because *S. alterniflora* is less resistant to decomposition.

The sites differed in the amounts of *S. patens* and *S. alterniflora they* contained. Bertness (1991) showed that when these two species occur together, *S. patens* dominates less flooded sites by out growing *S. alterniflora*, and *S. alterniflora* dominates more flooded sites that *S. patens* cannot tolerate. Thus the differences between the sites in species composition indicates that the more saline site was more flooded than the less saline site, which agreed with the Eh observations. Given that *S. alterniflora* was not limited by *S. patens* at the more saline site, an unanswered question was what limited *S. alterniflora* at the more saline site.

Aboveground production was greater than that generally reported for *S. patens* on the Atlantic coast of the United States, but only about half that

generally reported on the Gulf Coast (Table 7). Also, Pezeshki et al. (unpublished manuscript in review) found lower *S. patens* carbon assimilation rates in this area than at other Louisiana brackish marsh areas. Belowground production at both sites was greater than the 3 estimates by Gallagher and Plumley (1979) on the Atlantic coast, but less than that reported in Massachusetts by Valiela et al. (1976) (Table 7). Belowground production was 1.1 and 0.9 times aboveground production at the less saline and more saline sites, respectively. In a Georgia *S. alterniflora* marsh, belowground production was 1.6 times aboveground production (Schubauer and Hopkinson 1984), but I am unaware of similar estimates from Gulf Coast or other *S. patens* marshes.

I expected respiration would be less than previously reported for brackish marsh because of the frequent flooding at these sites. However, the previous estimate of respiration in brackish marsh (from data in DeLaune and Smith 1984) was similar to that at the less saline site, and less than that at the more saline site (Table 7). It should also be noted that respiration may be slightly underestimated. I observed many crustaceans and small fish on the marsh surface. Their density regurlarly appeared high, but I did not sample with any enclosed in the chambers.

The greater respiration at the more saline site likely resulted from the greater amount of *S. alterniflora* there. Soil respiration is greater in saline marsh soil than in brackish marsh soil (Chapter 4), most likely because *S. patens* is more resistant to decomposition than *S. alterniflora* . In fact, organic matter lost through respiration at the more saline site was more similar to that previously reported for saline marsh, 728 g m^{-2} yr^{-1} (from data in Feijtel et al. 1985), than to that reported for brackish marsh .

Table 7. Organic matter fluxes previously reported in *Spartina patens* dominated marshes, with emphasis on Gulf of Mexico marshes.

production ($g\ m^{-2}\ yr^{-1}$)	location	source
	aboveground	
1,341	Louisiana	White et al. (1978)
1,945	Louisiana	DeLaune and Smith (1984)
4,159	Louisiana	Hopkinson et al. (1980)
6,043	Louisiana	Hopkinson et al. (1978)
	belowground	
2,520	Massachusetts	Valiela et al. (1976)
120	Georgia	Gallagher and Plumley (1979)
150	Delaware	Gallagher and Plumley (1979)
220	Maine	Gallagher and Plumley (1979)
	respiration	
436	Louisiana	DeLaune and Smith. (1984)
	burial	
510	Louisiana	DeLaune and Smith (1984)
	export	
998	Louisiana	data in DeLaune and Smith (1984)

Burial was greater at these sites than at other sites for which estimates are available (Table 7). This likely resulted from the rapid belowground production in hummocks, much of which was apparently buried and subsequently elevated the hummocks above the surrounding soil that lacked rooted stems. However, vertical accretion was still inadequate to counter the estimated submergence rate of 1.38 cm/yr. This vertical accretion deficit is the most likely cause of the excessive marsh flooding, and was greater at the more saline site. If water levels continue to increase faster than the marsh builds up, then these marshes will gradually drown and convert to open water. A more extensive study of vertical accretion indicated that since 1964, this general area has lost at least 11 cm elevation relative to mean water levels (Chapter 7).

As noted, both sites were characterized by elevated hummocks surrounded by soil almost devoid of rooted vegetation. Hummock elevation appeared to result from vigorous root production by *S. patens* at culm bases immediately above the hummock surface, and subsequent silt capture. This may be a response to frequent flooding as Naidoo et al. (1992) noted that flooding stimulated production of negatively gravitropic roots in *S. patens*. Root growth, as well as production of upright stems, was also noted from decumbent stems of *S. patens*, but they never persisted. Their failure to persist might result from salinity fluctuations; Naidoo et al. (1992) noted that production of negatively gravitropic roots was inversely related to salinity. *S. patens* growth at the more saline site was never great enough to form large mats like those at the less saline site. I noted at Marsh Island, Louisiana that such mats may exceed 10 cm in thickness and give rise to persistent, upright stems, but that apparently requires more drying, lower salinity, or both, than

occurs at these sites. I have occasionally noted layers of horizontal stems in cores collected in brackish marsh for ^{137}Cs dating. Thus although root production appears to be most important for soil formation, such mat production may also partly counter submergence in *S. patens* marsh. *S. alterniflora* also exhibited some adventitious root growth from culm bases at both sites, but never formed floating mats.

The bare areas between hummocks were observed to be flooded almost permanently. The soil conditions expected to result from such flooding are the most likely reason that these areas did not support rooted vegetation, and were not colonized by new plants during this study. Drainage of the low areas between hummocks was observed only during prevailing north winds that occur during winter and spring and that cause extremely low tides. At normal low tides, the surface of most of hummocks were barely exposed, and the surface of bare soil was flooded by 10-20 cm of water. Plants rooted in hummocks were thus protected somewhat from flooding stress by their elevation. Plants rooted in the hummocks may have been further protected from anoxia by the extremely dense root network and lack of dead material in the hummocks; which might partially counter anoxia via the loss of relatively oxidized substances (Chalmers 1982).

Previous field studies indicated that marshes can export organic matter, including dissolved organic carbon (Craft et al. 1989 and articles cited therein). But these estimates were greater than generally reported and more similar to modeling results that yield exports of 1,000 g of C per m2 per year (Mitsch and Gosselink 1984:206). This may be because these study sites were dominated by *S. patens*, but previous studies have generally been made in *S. alterniflora* marsh, which are less productive (Pezeshki and DeLaune 1991), or

because previous studies usually omitted belowground production from their budgets (Feijtel et al. 1985), or assumed that belowground production equaled that buried (e.g. Day et al. 1973), and assumed that respiration recycled only aboveground production. Although those assumptions may not be accurate, these data provide little guidance as to how these generalization should be made when they are necessary. At the less saline site, it appeared that burial and soil respiration accounted for all belowground production. This would mean that nearly all aboveground production was exported to the surrounding shallow lakes. This is not likely because as noted, I noticed many small crustaceans and fish that appeared to be grazing among dead stems and leaves floating on the water. At the more saline site, it appeared that respiration involved the recycling aboveground production as well as belowground production. A brackish marsh on the Mississippi coast dominated by *Juncus* imported organic matter (Hackney and de la Cruz 1982). An important difference between their site and these sites is that their sites were not regularly flooded, and litter built up greatly on the marsh floor until it was removed by fire or hurricanes.

Mineral sediments provide nutrients for salt marsh vegetation (DeLaune et al. 1981), and plant biomass is related to soil nutrient content in Gulf Coast and Atlantic coast salt marshes (Broome et al. 1975, DeLaune and Pezeshki 1988). Soil bulk density depends on mineral sediment content (Hatton et al. 1983), thus plant biomass is related to soil bulk density in Louisiana salt (DeLaune and Pezeshki 1988) and brackish marshes (Chapter 10). In addition to nutrients, soil mineral matter contains Fe, which likely buffers against sulfide toxicity (King et al. 1982). However, no relationship was detected between biomass of either of these species and soil bulk density in

this area within the range sampled (unpublished data), which suggests that factors other than nutrient availability, such as flooding, regulated plant growth. Regardless, average soil bulk density at these sites, 0.09 g cm^{-3}, is too low to support healthy *S. alterniflora* marsh, which appears to require at least 0.25 g cm^{-3}(DeLaune et al. 1990). Thus it does not appear that these *S. patens* dominated marshes will convert to *S. alterniflora* dominated marshes in response continuing increases in flooding, salinity, and sulfate unless mineral sedimentation also increases so that soil bulk density increases. Insufficient soil mineral matter for brackish marsh to convert to saline marsh partly explain the large area of open water where brackish marsh previously existed SW of the more saline site.

CONCLUSIONS

All indications were that flooding was excessive in these marshes because of greenhouse-like conditions brought on by subsidence. Excessive flooding limited production and subsequent organic matter burial and export. Export is important because it links the marsh to the rest of the estuary; burial is important because it is required for marsh vertical accretion. Excessive flooding was both a cause and an effect of inadequate production because of the dependence of marsh elevation on organic matter burial, and the dependence of plant production on marsh elevation. These negative effects of greenhouse-like conditions on production appeared more pronounced where marine intrusion was more advanced, thus vertical accretion and export were lower there. It is also possible that *S. alterniflora* may not succeed *S. patens* in these mineral poor soils as marine intrusion and flooding progress. Although recent evidence indicates that much saline marsh production is

respired internally, these data indicated that tidal, non-saline marsh can export carbon when flooded often, as mangrove swamps that export large amounts of organic matter to surrounding water bodies (Boto and Bunt 1982). Further work is needed to investigate this possible export function of tidal, non-saline marshes.

CHAPTER 7

SOIL FORMATION IN A RAPIDLY SUBMERGING COASTAL MARSH

INTRODUCTION

The study in the previous section documented that vegetation was stressed as expected in a deteriorating marsh. It also appeared that this stress resulted primarily from flooding, and perhaps from inadequate soil mineral matter. If flooding caused of the plant stress, then that would mean that marsh vertical accretion was inadequate. Purposes of this study were to determine if vertical accretion was adequate in the study area and if so why. This required that the relative importance of organic matter and mineral matter to marsh vertical accretion be determined.

METHODS

Fifteen sites were sampled for analyses of soil formation (Figure 2). Streamside marsh was not sampled; all sites were located in inland marsh, i.e., beyond 5 m from the edge of bayous and lakes. Three saline marsh sites were sampled in August 1989, others were sampled between February 1990 and August 1990. The 4 most northern sites sampled in 1990 were in brackish/saline marsh; others were in saline marsh. Two soil cores were collected from each site. Cores were 45-55 cm long and were collected with 15-cm diameter, thin-walled, sharpened, aluminum, irrigation pipe. Cores were sectioned into 3 cm increments and dried at 80 ^{o}C to remove non-structural water.

Total vertical accretion on the marsh surface since 1963 was estimated from the depth of the soil increment containing the ^{137}Cs maxima, and the

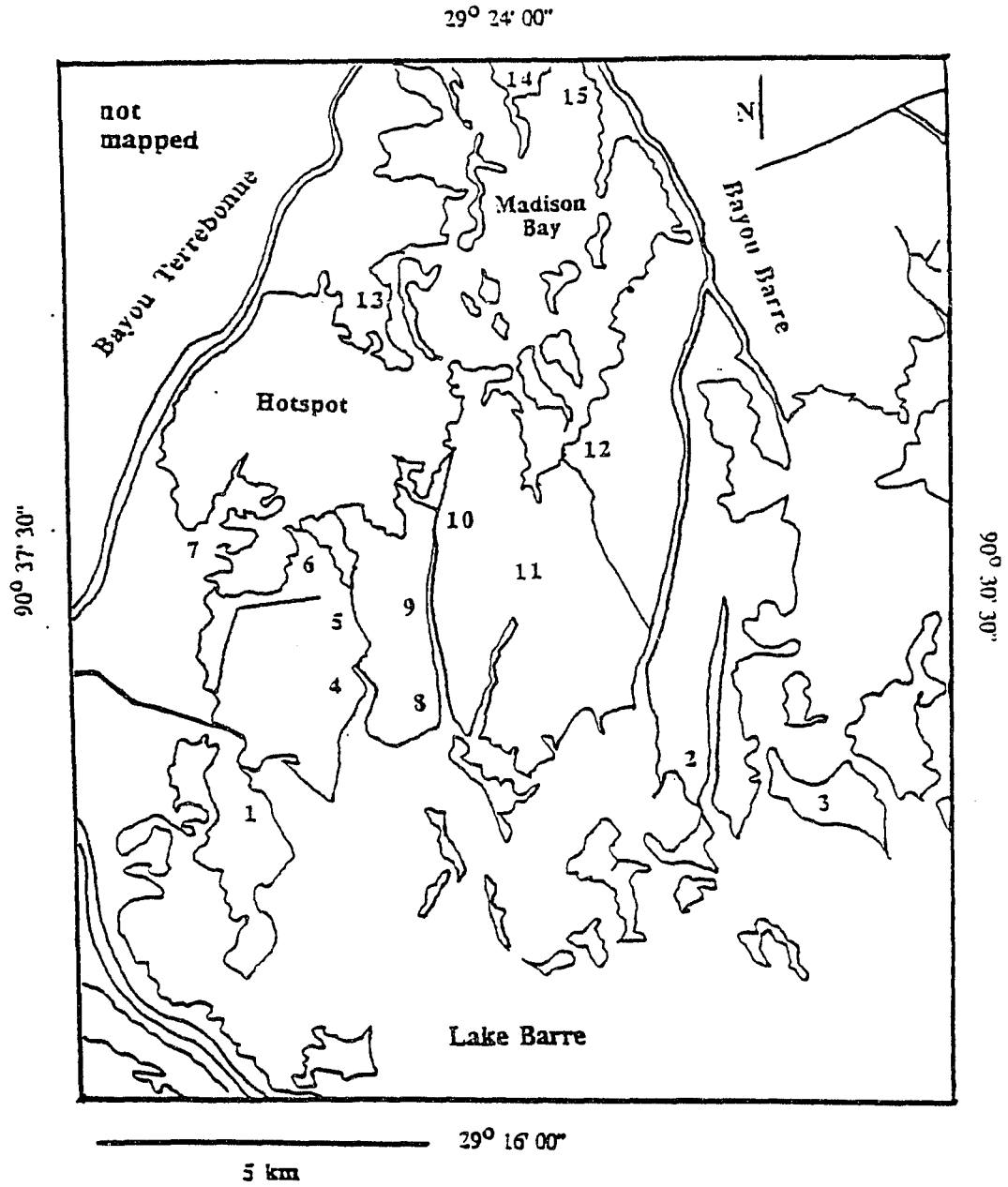

Figure 18. Approximate location of sites where replicate cores were collected for study of soil formation in the Lake Barre study area.

number of years between core collection and 1963 (DeLaune et al. 1978). See pages 62-63 for a detailed description of this technique. Vertical accretion rates were compared to published submergence estimates since 1963 to determine if vertical accretion in the study area was adequate to counter submergence. These local submergence rates more accurately reflect water level increases in the study area than global sea level rise do because the local rates also reflect local subsidence, which is substantial in coastal Louisiana (Penland and Ramsey 1990).

Initial efforts also included ^{210}Pb dating techniques to estimate vertical accretion (Armentano and Woodwell 1975), but were discontinued when I found that these cores were not long enough to reach beyond depths containing atmospherically deposited ^{210}Pb (unpublished data). Evidently, vertical accretion rate was so rapid that these cores (about 50 cm long) were too short to reach soil 80 years old. This information was desired because it would have provided information relevant to autocompaction.

Bulk density in the study area was determined from the weight of oven-dried soil increments, and the volume of the increments prior to drying. Bulk density estimates from the upper 5 increments (15 cm) of each core were used to estimate root zone soil bulk density at the 15 sites throughout the study area.

The amount of organic matter and mineral matter that accumulated since 1963 in each core was also estimated. Soil increments that contained the 1963 marsh surface, and all overlying increments were ground for analyses. Samples from each soil increment were ignited at 400 ^{o}C for 16 hr to remove organic matter (Davies 1974). The weights of mineral and organic matter per cm^3 of soil in each increment were estimated from soil bulk density,

percent mineral, and percent organic matter measured individually for each soil increment. Volumes of mineral and organic matter in soil were estimated as described in Chapter 3. Total mineral and organic matter accumulation could not be estimated for 2 cores from different sites because data were missing from individual increments in those cores.

Vertical accretion, mineral matter and organic matter accumulation rates since 1963 and soil bulk density within the root zone were compared between brackish/saline and saline marsh as a Completely Randomized Design with nesting of site within marsh type (Proc GLM, SAS Institute Inc., Cary, North Carolina, USA); the site-within-marsh-type (n = 15) term was the proper estimate of error for these tests rather than the core-within-marsh-type (n = 30). All soil increments in the upper 15 cm of soil (n = 148) were used to estimate soil bulk density in the study area, but to be conservative, soil increments were considered sub-samples and were not used as independent estimates of error for statistical tests. Rather, the site-within-marsh-type term (n = 15) was used. Vertical accretion and mineral and organic matter accumulation were analyzed (Proc GLM, SAS Institute Inc., Cary, North Carolina, USA) to determine whether organic matter accumulation, mineral matter accumulation, or their interaction was most strongly related to vertical accretion. For these regressions, type III "uniquely attributable" sums of squares were used. This is especially important because mineral and organic matter accumulation were expected to be highly correlated, i.e., these data were expected to be ill-conditioned. Vegetation and soil bulk density data were analyzed (Proc REG, SAS Institute Inc., Cary, North Carolina, USA) to test for a linear relationship between soil bulk density and saline marsh

vegetation above-ground biomass within the study area. For all tests, an alpha level of 0.05 was used.

RESULTS

The 1963 marsh surface was almost always located at identical or adjacent depths in replicate cores from the same site (e.g., Figure 3). Vertical accretion since 1963 in the study area averaged 0.98 cm/yr (n = 30, SE = 0.06), and was similar in brackish/saline and saline marsh ($F = 0.03$, 1 and 13 df, $P = 0.8609$) (Table 8). There was no obvious spatial pattern in vertical accretion rates. Brackish/saline and saline marsh also had similar organic matter accretion rates ($F = 0.14$, 1 and 13 df, $P = 0.7111$), but differed greatly in mineral matter accumulation ($F = 5.95$, 1 and 13 df, $P = 0.0298$) (Table 8).

Whereas ^{137}Cs profiles represented continual atmospheric deposition, variable soil bulk density with depth reflected the discontinuous process of mineral sedimentation (e.g., Figure 19). Soil bulk density within the root zone, the upper 15 cm of soil, was lower in brackish/saline marsh than in saline marsh ($F = 22.11$, 1 and 13 df, $P = 0.0004$).

Vertical accretion was correlated with both mineral matter ($R = 0.70$, n = 28, $P = 0.0001$) and organic matter accumulation ($R = 0.95$, n = 28, $P = 0.0001$). These Pearson correlation coefficients indicated that the linear relationship between vertical accretion and organic matter content was stronger than the linear relationship between vertical accretion and mineral matter (Figure 20). Mineral matter accumulation and organic matter accumulation were also correlated ($R = 0.75$, n = 28, $P = 0.0001$), thus these data were ill-conditioned as expected. Test of hypotheses with type III sums of squares indicated that vertical accretion was related to the amount of

Table 8. Soil formation characteristics in the Lake Barre area marshes, data from two cores at each site, Terrebonne Parish, Louisiana, 1989 and 1990.

site	bulk density† (g/cm^3)	organic matter (%)	vertical accretion (cm/yr)	mineral accum. ($g\ m^{-2}yr^{-1}$)	organic accum. ($g\ m^{-2}yr^{-1}$)
			---------- saline ----------		
1. Lake Barre	0.25	31.5	1.78	4,178	1,245
2. Charles Theriot	0.27	17.4	0.98	2,083	440
3. Bay la Peur	0.31	24.4	0.78	2,053	567
4. downstream Chitigue	0.27	21.2	0.98	2,341	576
5. midstream Chitigue	0.26	21.9	0.75	1,639	456
6. upstream Chitigue	0.21	29.9	1.22	1,928	720
7. Grand Bayou‡	0.18	31.0	1.04	1,126	406
8. dwnstream deMangue‡	0.20	39.5	0.56	1,376	439
9. midstream deMangue	0.26	23.5	1.28	2,643	776
10. upstream deMangue	0.17	29.6	0.94	1,100	624
11. DuFrene	0.16	31.3	0.55	699	310
mean: saline	0.23	27.4	0.99	1,992	613
			------ brackish/saline ----		
12. S.E. Madison Bay	0.13	47.1	0.67	513	434
13. W. Madison Bay	0.09	44.5	0.78	394	345
14. N. Madison Bay	0.13	45.3	1.33	1,054	796
15. N. Billy Goat Bay	0.13	38.9	1.06	934	592
mean: brackish/saline	0.12	44.0	0.96	724	542
			------ mean of all sites ----		
	0.20	31.8	0.98	1,629	593

† soil bulk density in the root zone only (top 15 cm of soil); all other parameters estimated from all soil overlying the 1963 marsh surface

‡ mineral and organic matter accumulation rates could be determined in only one core

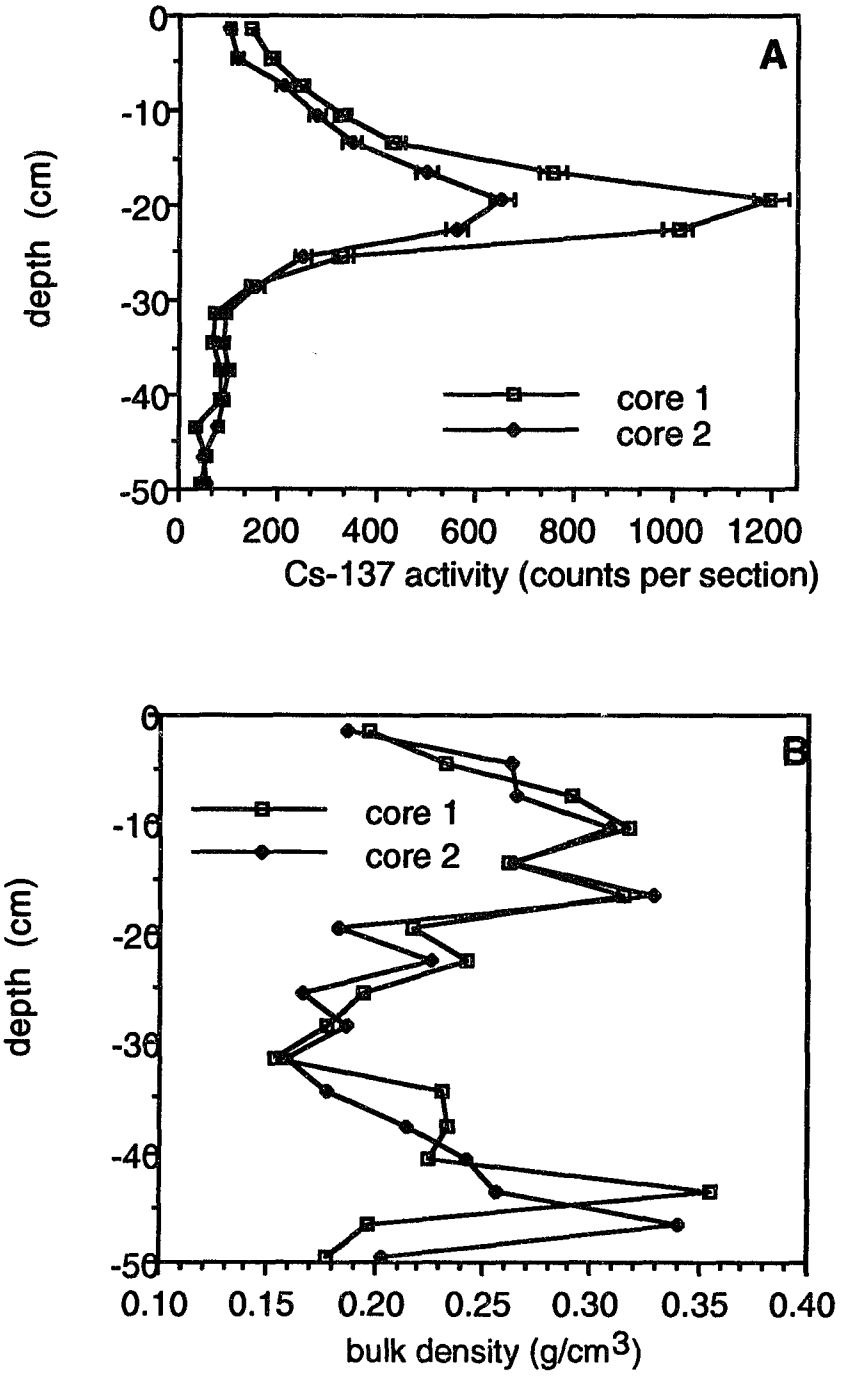

Figure 19. ^{137}Cs profiles and soil bulk density profiles in replicate cores from a site in the Lake Barre study area. ^{137}Cs activity reflects continual atmospheric deposition, with peak levels in 1963. Soil bulk density reflects the discontinuous process of mineral sedimentation.

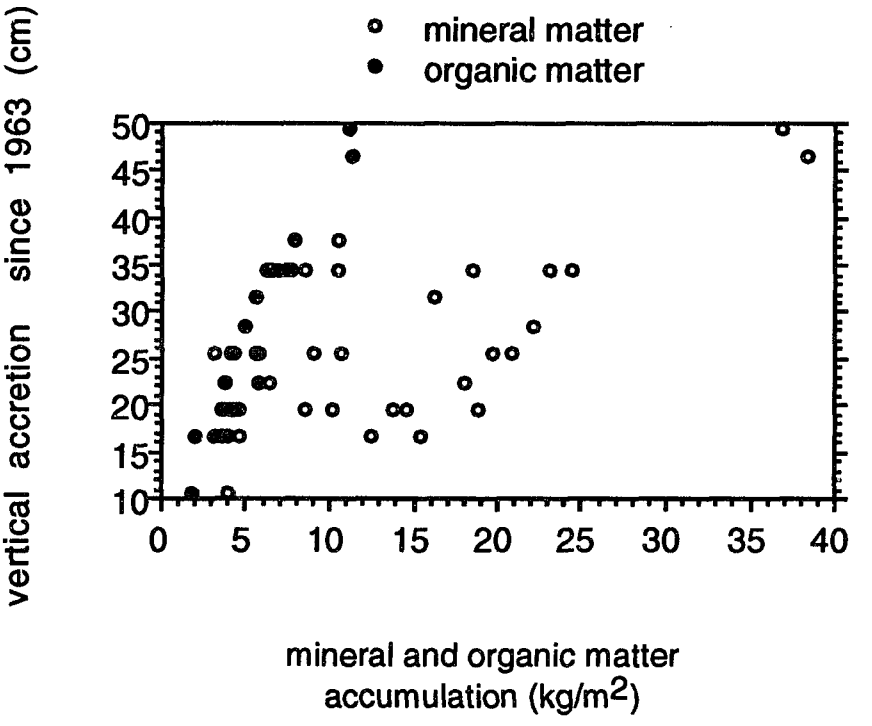

Figure 20. Relationships between vertical accretion since 1963 and the amounts of mineral and organic matter accumulated since 1963 in the Lake Barre/Madison Bay study area, 1989/1990.

organic matter accumulated (F = 52.12, 1 and 23 df, P = 0.0001), but not to the amount of mineral matter accumulated (F = 0.01, 1 and 23 df, P = 0.9696). The interaction between mineral matter and organic matter was not significant (F = 0.05, 1 and 23 df, P = 0.8248). This analyses indicated that variation in vertical accretion that could be explained by mineral matter accumulation could also be explained by organic matter accumulation, but the reverse was not true. Partial plots (Proc REG, SAS Institute Inc., Cary, North Carolina, USA) were also generated to test this interpretation of these ill-conditioned data. They indicated no relationship between mineral matter accumulation and vertical accretion after adjusting for organic matter accumulation. However, a very strong linear relationship still existed between organic matter accumulation and vertical accretion after adjusting for mineral matter accumulation.

DISCUSSION

Vertical accretion since 1963 was very high; at these sites it generally exceeded the rate for brackish and saline marshes in southeast Louisiana (0.72 cm/yr, Chapter 3). Even though vertical accretion in the study area was greater than elsewhere, it was inadequate to counter submergence. As noted earlier, estimated submergence since 1964 at 6 U.S. Army Corps of Engineers tide gauge stations surrounding the study basin averaged 1.38 cm/yr, or 41% greater than vertical accretion, and that was a conservative estimate (Table 4). The vertical accretion deficit would cause the marsh surface to lose 11 cm of elevation relative to mean water levels since 1964 even neglecting compaction below the 1963 marsh surface. This is quite large relative to average marsh

elevation in Louisiana's flat marshes (about 30 cm above mean water levels, (Chabreck 1970:12)), and would increase flooding stress on vegetation.

Limited field observations also indicated that there was a vertical accretion deficit. During summer, I observed that the marsh surface did not completely drain even at extreme low tide. I observed marsh drainage only during winter when persistent north winds lowered water levels for extended periods of time. Continuous water-level data collected in the study area over an 11 day period during October 1990 indicated that, although there was a high and low tide each day, the marsh surface drained on only 7 days and only very briefly; the marsh surface was flooded more than 90% of the time during the 11 day period (from data in Reed et al. 1991 unpublished Open File Report to United States Geological Survey, grant no. 14-08-0001-23320).

The vertical accretion deficit and resulting submergence, rather than lateral erosion, appeared to be the mechanism by which marsh converted to open water at the hotspot. There was no distinct interface between the hotspot and surrounding marsh; instead, hummocks of vegetation gradually gave way to open water with no obvious change in elevation. No evidence of lateral erosion was noted and plant stubble was still rooted in place beneath the water at areas that converted from marsh to open water during 1990 and 1991.

Vertical accretion must result from organic matter accumulation, mineral matter accumulation, or a combination of both. However, statistical analysis indicated that variation in vertical accretion among these sites was significantly related only to variation in organic matter accumulation among the sites. Apparently, differences in mineral sedimentation rates among the sites resulted primarily in differences in soil bulk density. Thus I concluded that

vertical accretion in the study area depended directly on organic matter accumulation. I am unaware of other quantitative analyses to compare to these results, but others also concluded that organic matter accumulation controlled vertical accretion rates in Louisiana (Hatton et al. 1983) and New England marshes (Bricker-Urso et al. 1988, McCaffrey and Thomson 1980, Stumpf 1983). This does not likely apply to mineral soil marshes such as those on the Atlantic coast of the southeastern United States.

Given that vertical accretion depended on organic matter accumulation, inadequate vertical accretion resulted from inadequate organic matter accumulation. Inadequate organic matter accumulation could result from either excessive decomposition, inadequate belowground plant production, or a combination of both. However, excessive flooding such as occurs in the study area reduces both decomposition (Chapter 4), and production in *S. alterniflora* and *S. patens* marsh (DeLaune et al. 1983b, Mendelssohn and McKee 1988, Pezeshki et al. 1991). Thus inadequate organic matter accumulation most likely resulted from inadequate production. Evidently, decreases in soil organic matter decomposition rates and increases in root/shoot ratios that are also expected with increased flooding (Good et al. 1982, Chapter 4) were not adequate. Given that plant production was limited by flooding, and that vertical accretion depended on organic matter accumulation, then a positive feedback loop of inadequate vertical accretion and plant flooding stress is indicated.

It would be incorrect to conclude from these data that mineral matter accumulation was unimportant to the vertical accretion process even though vertical accretion depended directly on organic matter accumulation. Organic matter accumulation probably depends partially on mineral matter

accumulation because, as noted, soil mineral matter enhances plant growth. This is the most likely reason that organic matter accumulation and mineral matter accumulation were positively correlated. Others have also noted that organic matter accumulation and mineral matter accumulation were positively correlated, and concluded that this resulted from the positive effect of mineral sediments on plant growth (Bricker-Urso et al. 1989).

Brackish/saline marsh in this study area accreted vertically at rates similar to the saline marsh, but incorporated only 40% as much mineral matter as saline marsh. The difference in mineral matter accumulation between brackish and saline marsh may result from differences in mineral matter availability (e.g., McCaffrey and Thomson 1980) or because brackish and saline marshes differ in SO_4^{-2} availability. This would cause them to have different mineral matter requirements to counter different levels of sulfide toxicity (Chapter 3).

Mineral and organic matter accumulation rates were inadequate to offset submergence (Table 9: required value). Requirements were estimated from the submergence rate in this study area and typical soil properties, according to the equations generated in the first study. For example, one vertical cm of brackish marsh soil usually contains 583 g of organic matter and 1,052 g of mineral matter for each square meter of marsh. Thus 1.38 cm/yr * 583 g/m^2 cm^{-1} = 804 g m^{-2} yr^{-1} of organic matter and 1,452 g m^{-2} yr^{-1} of mineral matter were required at these brackish sites. It is important to note that the accumulation rates were not low relative to other marshes (Table 9), thus even typical plant production may be inadequate during rapid submergence. Apparently, vertical accretion is self-regulating only during moderate submergence. Organic matter accumulation has an upper limit set

Table 9. Vertical accretion and mineral and organic matter accumulation rates measured at this site (Lake Barre, Louisiana), required† at this site, and measured at other coastal marshes.

source	vertical accretion (cm/yr)	accumulation rates ($g\ m^{-2}\ yr^{-1}$) organic matter	mineral matter
		brackish marsh	
measured	0.96	542	724
required	1.38	804	1,452
Barataria Basin‡	0.59	348	478
		saline marsh	
measured	0.98	613	1,992
required	1.38	829	2,481
Barataria Basin	0.75	435	1,740
Chesapeake Bay¥	0.78	775	1,859

† soil formation requirements estimated from the submergence rate and equations in Chapter 3, figure 4)
‡ Hatton et al. (Hatton et al. 1983, table 1)
¥ Kearney and Stevenson (1991, table 2)

by production; mineral matter accumulation has an upper limit set by mineral matter availability and the frequency of sedimentation events.

CONCLUSIONS

Marsh in this study appears threatened by a hypothetical positive feedback loop of plant flooding stress and inadequate vertical accretion. Vertical accretion and plant growth are related because plant growth is a function of flooding, which depends partly on marsh elevation, and vertical accretion depends partly on organic matter accumulation (Figure 21). Vertical accretion in this study area depends directly on organic matter accumulation, but plant growth supplies inadequate soil organic matter to counter submergence. Thus, inadequate plant productivity may increase the vertical accretion deficit, initiate another cycle of declining plant production and increasing vertical accretion deficits (Figure 21). There is no way to determine which came first: inadequate vertical accretion or inadequate organic matter accumulation. Changes in the subsidence rate and ephemeral plant stresses, such as high water or cyclic herbivory, are the most obvious triggering factors.

An important question is what prevents the negative feedback system that commonly maintains marsh elevation from replacing this hypothetical positive feedback system (Figure 21)? Although increased tidal flooding usually increases mineral matter accumulation (Mitsch and Gosselink 1986:178), tidal flooding is relatively unimportant to sedimentation in coastal Louisiana and winter storms are more important (Reed 1989). Thus, the most important sedimentation events probably have not greatly increased in frequency even though the marsh is flooded much of the time. The failure for tidal sedimentation to greatly increase and the low sediment availability in the

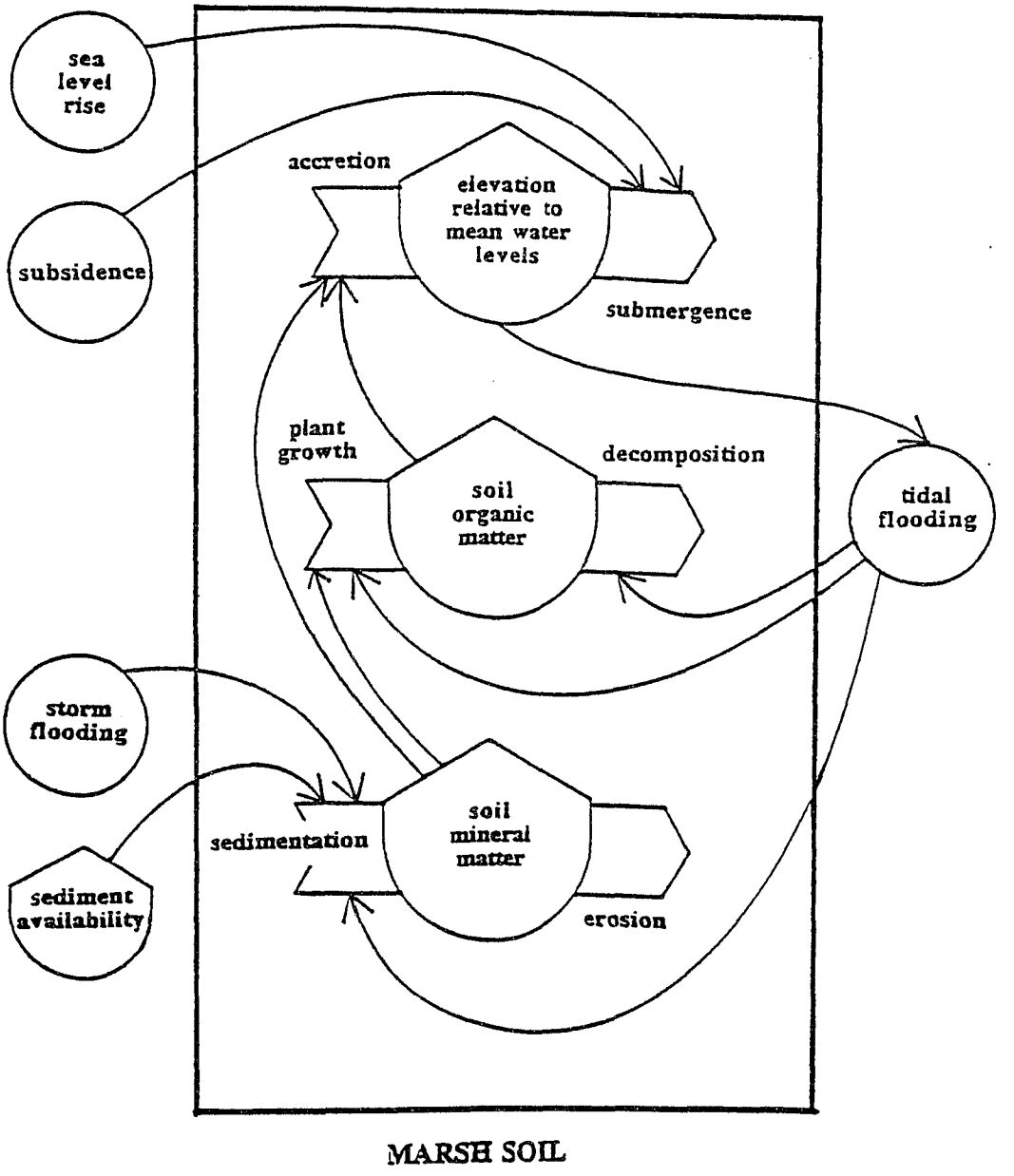

Figure 21. Conceptual model of hypothesized interactions among soil formation processes during submergence. Vertical accretion can be adequate when increased flooding decreases soil organic matter decomposition and increases sedimentation. Vertical accretion can be inadequate when flooding limits plant production and vertical accretion depends directly on organic matter accumulation. In this case, inadequate plant growth increases the vertical accretion deficit, which promotes another cycle of declining plant growth and increasing vertical accretion deficits.

area are the most likely reasons that the more common negative-feedback system has not replaced the hypothetical positive feedback system. Additional information regarding the relative importance of mineral and organic matter accumulation is needed to better understand marsh soil formation. Even though the situation indicated in this study area is not likely common in areas with typical subsidence rates, such information is required to determine if similar positive feedback situations might develop in other coastal marshes as a response to possible increases in the rate of global sea-level rise.

SECTION IV: TEST OF HYPOTHESIS AT MARSH ISLAND

CHAPTER 8
FIELD STUDIES AT MARSH ISLAND

INTRODUCTION

The hypothesis developed as a result of the field studies at Lake Barre needed to be tested in another area experiencing marsh loss. Marsh Island, Louisiana was chosen as the study site because the rates of marsh loss were previously documented and the areas where marsh loss occurred were known (Nyman et al. 1990).

Aerial photographs indicated that marsh loss at Marsh Island occurred via the expansion of interior broken marsh at the expense of surrounding unbroken marsh (Nyman et al. 1990). Such broken marsh surrounded by solid marsh is the most common marsh landform in Louisiana. Turner and Rao (1990) used 72 7.5-minute quadrangle maps from throughout the Louisiana coastal marshes to examine wetland loss that occurred between 1956 and 1978. They found that fragmentation of the marsh interior was much more prevalent than shoreline erosion. There is no canal and spoil bank network on Marsh Island, thus, marsh loss is not likely related to salt water intrusion or accidental impoundment. Instead, interior breakup was hypothesized to be related to naturally occurring hydrologic isolation of the marsh interior, i.e., insufficient mineral sedimentation for marsh vertical accretion. The resulting low areas would impound water and limit plant production, which would limit vertical accretion and cause ever increasing vertical deficits and flooding similar to what appears to occur in the Lake Barre area (see chapter 7).

STUDY AREA

Marsh Island is a 31,000 ha island on the central Louisiana coast (Figure 22). Underneath are 10-17 m of clays, peats, and sands that are primarily the products of previous Mississippi River distributaries (Orton 1959). The island is undeveloped and is entirely occupied by the Marsh Island Wildlife Refuge, operated by the Louisiana Department of Wildlife and Fisheries. Some larger lakes and bayous have 30-cm natural levees, but otherwise the marsh is flat km after km (Orton 1959). For the most part, bayous drain north into the brackish bays rather than south into the Gulf of Mexico, and for at least the last 40 years, dominant vegetation has been *S. patens* and either *Scirpus olneyi* Gray or *Juncus roemerianus* Scheele (Chabreck and Condrey 1979, O'Neil 1949, Orton 1959). There is a large impoundment and a large area under the influence of weirs to attract wintering waterfowl to a refuge from hunting, but most of the island is unmanaged other than prescribed burns for the purpose of promoting *Scirpus olneyi* (see Chabreck and Condrey 1979 for a checklist of common vascular plants).

Although long abandoned by riverine processes and now dominated by marine processes, vertical accretion averaged 0.70 cm yr^{-1} between 1963 and 1985 (DeLaune et al. 1987). Unlike many coastal marshes in Louisiana, the island is relatively unaffected by canals. Marsh loss occurs, but at a relatively modest rate. The landscape pattern is typical for coastal Louisiana (Turner and Rao 1990b): interior broken marsh that expands at the expense of surrounding unbroken marsh. Study of aerial photographs made of Marsh Island in 1957 and 1983 indicated that marsh loss is relatively moderate and proceeds via the conversion of unbroken marsh to broken marsh at a rate of 0.37% yr^{-1}, or approximately 115 ha yr^{-1} for the entire island (Nyman et al.

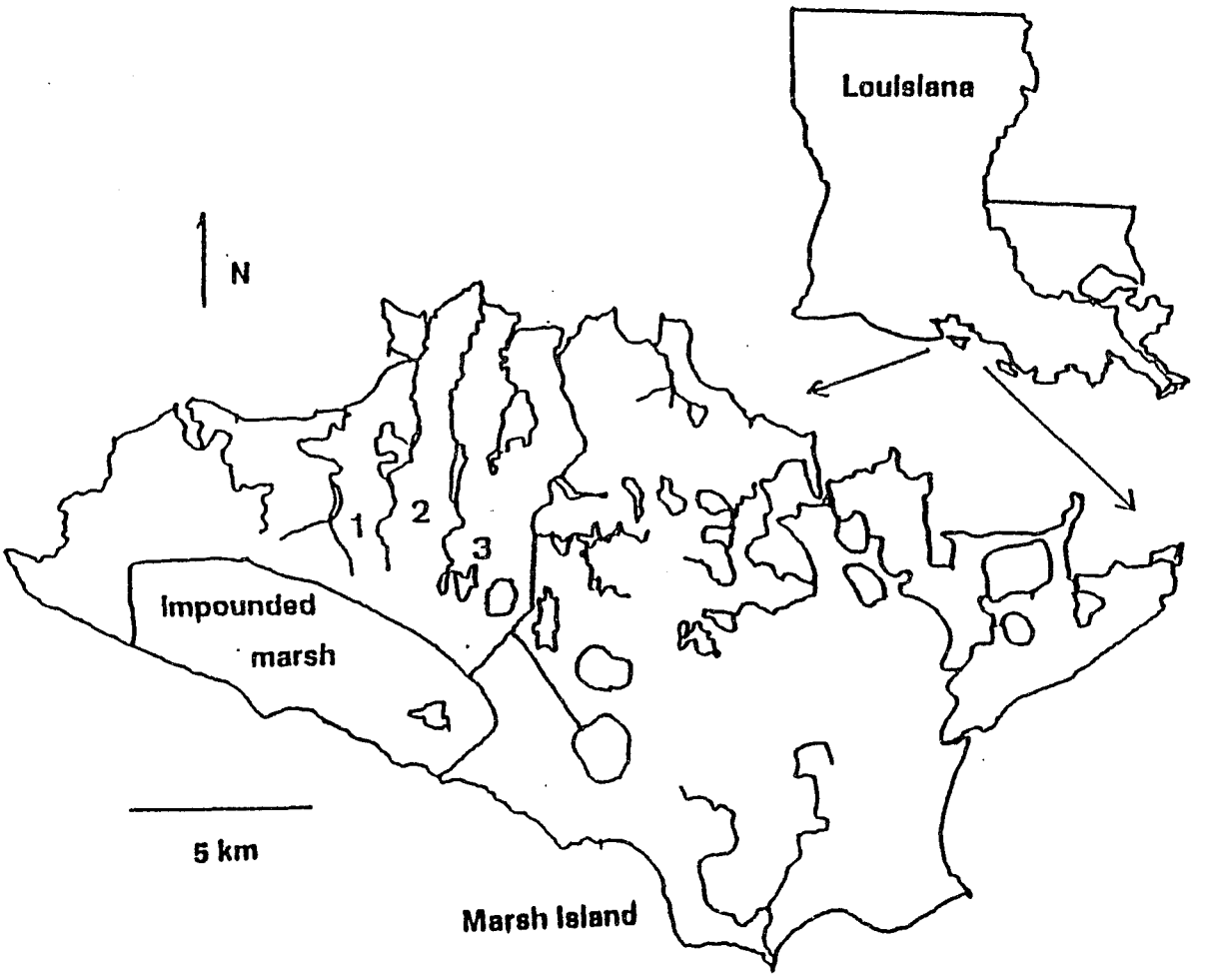

Figure 22. Marsh Island, Louisiana, where field work was conducted. Numbers 1,2, and 3 denote the location of broken marsh/solid marsh complexes sampled in 1990.

1990). Marsh loss rates in this region of Louisiana were greater between 1956 and 1974 than between 1974 and 1983 (Britsch and Kemp 1991).

CHAPTER 9

A PRELIMINARY COMPARISON OF VEGETATION AND SOIL IN HEALTHY AND DETERIORATING MARSH

INTRODUCTION

There were 2 purposes of this field work. The primary purpose was to determine if plant stress and wetland loss were related as had been observed in the Lake Barre area (Chapter 6). My initial hypothesis was that the broken marsh areas in the marsh interior functioned as shallow bowls. During high rainfall periods, they held water that stressed vegetation. During low rainfall periods, they concentrated salts that stressed vegetation. Furthermore, these interior broken marsh areas were likely to be starved for mineral sediments because they were so far from the large bayous and lakes. I therefore expected that broken marsh would be characterized by less robust vegetation resulting from waterlogging stresses, and lower soil bulk density resulting from isolation from sediment sources. Future studies would document the causes of stress. A secondary purpose was to search for soil mineral matter/plant biomass relationships in *Spartina patens* marsh. A similar relationship had already been established for the salt marsh species *Spartina alterniflora* (DeLaune and Pezeshki 1988).

METHODS

Two sites were sampled 25 October 1989; one in broken marsh and one in solid marsh (Figure 21). End-of-season biomass was harvested from 10 1-m^2-plots at each site. A sediment core, 30 cm in depth and 15 cm in diameter, was collected from the center of each harvested plot.

A water well was dug at each site by removing a 50 cm sediment core from the marsh. Water levels were determined almost simultaneously at the 2 sites 2 hours after the wells were constructed.

In the lab, the oven-dried weight of vegetation and sediment cores was measured to estimate end-of-season, standing-crop biomass and soil bulk density. Plant species that occurred at each site were noted to estimate species richness of each site. Soil cores were ground and mixed, and subsamples were taken to determine mineral matter and organic matter content of the core. Percent weight by mineral matter and organic matter were determined by loss-on-ignition at 400 ^{o}C for 16 hours (Ball 1964, Davies 1974). Percent by volume of mineral and organic matter was estimated from percent by weight and the estimates of the particle density of soil mineral matter (2.61 g/cm3) and soil organic matter (1.14 g/cm3) from DeLaune et al. (1983). Data were analyzed as 2-way ANOVA to test for differences in end-of-season standing-crop biomass, soil bulk density, mineral percent of soil volume, organic percent of soil volume, and soil pore space between the solid and broken marsh sites. An alpha level of 0.05 was used as the critical limit.

RESULTS

Vegetation appeared vigorous at both sites, but more vigorous at the broken marsh site. But it was also evident that the solid marsh site had burned the previous year, and litter was not common at the solid marsh site. All plots at both sites were dominated by *Spartina patens.* Three plant species occurred in plots harvested at the solid marsh site: *Spartina patens, Scirpus olneyi,* and *Juncus roemerianus.* At the broken marsh site, those

species occurred, as well as *Distichlis spicata, Vigna luteola, Lythrum lineare,* and *Aster subulatus.*

The solid marsh site had water on the marsh surface, and the water well was filled when revisited. At the broken marsh site, there was no water on the marsh surface, and the water in the water well was 18 cm below the marsh surface when revisited. This appeared to correspond to water levels in the adjacent pond area, but no measurements were made.

End-of-season, standing crop biomass averaged 1,608.9 g/m^2 at the broken marsh site, and 1,317.1 g/m^2 at the solid marsh site, but the difference was not significant ($F = 4.22$, df=1,18, $P = 0.0549$). Soil at the broken marsh site had a greater bulk density ($F = 38.88$, df=1,18, $P = 0.0001$) and a greater volume of mineral matter ($F = 21.02$, df=1,18, $P = 0.0002$) than soil at the solid marsh site (Table 10). There was 2.1 times more mineral matter in soil at the broken marsh site than at the solid marsh site. There was no difference between the sites in the volume of soil organic matter ($F = 0.02$, df=1,18, $P = 0.8906$) (Table 10). Soil pore space was lower at the broken marsh site than at the solid marsh site ($F = 7.53$, df=1,18, $P = 0.0143$).

DISCUSSION

Vegetation at the broken marsh site was expected to be less vigorous than vegetation at the solid site, but the data indicated otherwise. Unfortunately, because the solid marsh site had burned the previous winter, and the broken marsh site had not, planned comparisons of the vegetation between the 2 areas were not valid tests of differences between solid and broken marsh areas. Marsh Island biologists reported that fire did not spread across broken marsh, and those areas were in fact used as fire breaks to

Table 10. Soil characteristics 10 plots from solid marsh and 10 plots from broken marsh, Marsh Island, Louisiana, 1989.

characteristic	broken marsh		sold marsh	
	mean	SD	mean	SD
bulk density (g/cm^3)	0.16	0.02	0.10	0.01
% organic volume	5.0	3.5	4.7	0.6
organic density (g/cm^3)	0.06	0.02	0.05	0.01
% mineral volume	3.9	1.0	1.9	0.3
mineral density (g/cm^3)	0.10	0.02	0.05	0.01
% pore space	91.1	1.8	93.4	0.4

control the size of burns in solid marsh (M. Carloss, pers. comm.). Fire is used in brackish marshes to remove nearly monospecific stands of *Spartina patens* and to promote the growth of *Scirpus olneyi* (Lynch 1941, O'Neil 1949). Although the difference in biomass between the solid marsh site and broken marsh site was nearly significant, any possible difference between the 2 sites could have also been caused by the different burning histories, and these biomass data could not indicate if the sites exhibited different degrees of waterlogging induced plant mortality.

Previous workers have noted that species richness is generally lower in less-drained brackish marsh areas than in better-drained brackish marsh areas (Chabreck and Hoffpauir 1962). It was therefore expected that species richness would be lower at the broken marsh site than at the solid marsh site. The opposite was observed, and this difference was not likely the result of the previous burn at the solid marsh site because marsh fires generally increase, rather than decrease, species richness (Lynch 1941). Although not conclusive, the species richness of the two areas suggested that the solid marsh site was less well-drained than the broken marsh site, the opposite of what was expected.

Although the water wells were only visited once, they provided a means to compare the depth of soil drainage between the 2 sites during low tide. The solid marsh had water pooling on the marsh surface when it was visited, and water was apparent in the wells when they were dug. The broken marsh had no water on the marsh surface, and there was no water in the wells when they were dug. When the wells were visited 2 hours later, water in the wells at the solid marsh site was flush with the marsh surface, but water in the wells at the broken marsh site appeared level with the water

level in the surrounding pond area, 18 cm below the marsh surface. These observations excluded the possibility that the broken marsh site was less well drained than the solid marsh site. The tide was low when the site was visited, but not unusually so (M. Carloss, pers. comm.), and this observation was believed typical of low tide conditions. Horizontal movement of water in marsh soils in response to tidal action has been shown to be slight on the Atlantic coast (Agosta 1985, Gardner 1972, Hemond and Fifield 1982), and may also be slight in Louisiana marshes. The solid marsh area was apparently too far from a tidally influenced water body to experience soil drainage. However, the size of the marsh islets in the broken marsh area apparently allowed them to experience complete drainage in response to tidal drainage of the marsh ponds. The marsh ponds in the broken marsh area were all connected to a large tidally influenced bayou via a small bayou. This small bayou was also evident in 1957 aerial photographs, when the area of broken marsh was smaller. Thus the broken marsh was less isolated from tidally influenced water bodies than the solid marsh site, the opposite of what was expected.

The greater soil bulk density and amount of mineral matter at the broken marsh site indicated that the broken marsh site flooded more frequently with sediment laden waters than the solid marsh site. The mineral matter content was higher at the broken marsh site probably because the broken marsh consisted of islets surrounded by a tidally influenced pond area, and the additional mineral matter was delivered to the broken marsh site by tidal action. Stumpf (1983) found that 80 % of the suspended inorganic material in tidal flood water disappeared within 12 m of a marsh creek; and even the center of most marsh islets of the broken marsh area

appeared <12 m from the pond area. Recently, Stoddart et al. (1989) demonstrated the importance of drainage systems in distributing sediment across the marsh surface, and in saline marshes, differences in soil bulk density between streamside and inland marshes are attributed to differences in the distance between the areas and tidally influenced water bodies (DeLaune et al. 1979).

Vegetation at the broken marsh site may benefit from the greater soil bulk density at that site. Standing crop biomass in saline marshes is positively related to the volume of soil nutrients, which probably as a result of volume mineral matter in the soil (DeLaune et al. 1979). There appeared to be a relationship between soil bulk density and end-of-season standing crop biomass, but the soil bulk density effect was confounded with the site effect, and that relationship could not be tested.

CHAPTER 10
DETAILED COMPARISON OF SOLID MARSH AND INTERIOR BROKEN MARSH

INTRODUCTION

Because this pilot project indicated unexpected conditions, initial plans to investigate relationships among brackish marsh loss, plant stress, and inadequate soil mineral matter were modified so that a more detailed study of the vegetation and soil was conducted. The purpose of this field work was to firmly establish whether or not broken marsh vegetation was more stressed that solid marsh vegetation.

METHODS

Three unmanaged, broken marsh areas were identified on 1957 black and white aerial photographs (Figure 21). Comparison with 1983 color infrared aerial photographs indicated that these broken marsh areas had expanded into adjacent unbroken marsh in the interim, thereby causing marsh loss. Three transects were established 8 October 1990 in each area which originated near the center of the broken marsh, and which extended into unbroken marsh (Figure 22). These areas were not accessible by boat because of the extreme shallowness of the broken marsh water bodies, and an airboat was used to conduct all sampling. Three stations were established on each transect; the first was established in broken marsh 5 m from edge of the last water body encountered, the second was 30 m further into unbroken marsh, and the last was 90 m further into unbroken marsh. These distances were selected to insure that 1/3 of the 27 sampling stations were in marsh that

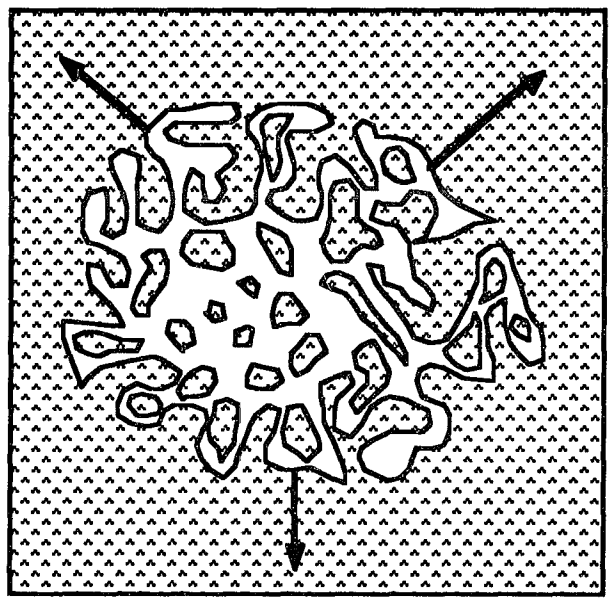

Figure 23. Diagram showing position of transects relative to broken and unbroken marsh. Transect starting points were in broken marsh at the edge of the water and extended inland as indicated by the arrows. There were 9 transects in all because 3 different broken marsh areas were sampled. There were 27 sampling stations in all because there were 3 sampling stations on each transect.

was obviously broken, another third were marsh that was obviously unbroken, and the remaining third were in transitional marsh.

Soil Eh can be used as an indicator of waterlogging stress on marsh vegetation (e.g. DeLaune et al., 1983a), thus soil Eh was compared between unbroken and broken marsh. Eh electrodes were installed at each station on one transect at 1 cm and 5 cm below the soil surface. Eh was recorded 8, 9, 10, and 11 October 1990.

Elevation on each transect was estimated from water depth measurements made almost simultaneously at all 3 stations within a transect. This required that the water levels be flat, i.e., that water levels were static. I believe this was true because these data were collected 9 October when water levels had been above the marsh surface for over 24 hours because of several days of strong south winds. Thus these measurements were used to determine the elevation of the 35-m and 125-m stations relative to the 5-m station, which was assigned an elevation of 0 cm within each transect.
Soil drainage was compared among stations. Water wells were constructed at each station 8 October 1990 by inserting a 50-cm long perforated PVC pipe 30 cm into the soil, and then removing the soil contained in the pipe. The pipe was 10 cm in diameter and perforated throughout the 30-cm portion inserted below the soil surface. Water levels were above the soil surface when the wells were constructed, but fell the night of 9 October because of strong N winds following a winter weather front. Water level in each well was recorded 10 October and 11 October, and the average was used as an index of soil drainage.

Above-ground biomass of marsh vegetation was compared between unbroken and broken marsh. Above-ground biomass was harvested from a

0.5 m2 plot at each station 9 October 1990. Burn stubble and low amounts of dead biomass was used as an indication of recent burning history, and plots were classified as (recently) burned and unburned. The oven dried weight of live and dead culms was determined for each species in each plot. The samples had to be held in a walk-in cooler 8 weeks before they could be sorted. Fortunately, only *S. olneyi* appeared to be affected to the extent that errors may have been made separating live culms from dead culms of this species. Thus, live and dead were combined for this species, and total above-ground biomass of this species was analyzed. Species richness was determined for each plot, and Simpson's dominance index was calculated for each plot as well (Simpson, 1949). Simpson's index is a measure of species diversity that ranges between 0 and 1, with values being greater in less diverse communities (Simpson, 1949).

Marsh vegetation biomass can be related to soil mineral matter content (Broome et al., 1975, DeLaune and Pezeshki, 1988, King et al., 1982), thus soil mineral matter content was compared between unbroken and broken marsh. A 15-cm diameter, 20-cm long soil sample was also collected from the center of each harvested plot. Soil samples were dried, and the bulk density was estimated from the dry weight and wet volume of the cores. Cores were then ground in a Willey mill, and subsamples were ignited at 400 ^{o}C to estimate percent organic matter (Davies, 1974). Soil mineral and organic matter density were then estimated from the soil bulk density and percent mineral and organic matter.

Vertical accretion rates were compared between unbroken and broken marsh because inadequate vertical accretion is associated with marsh loss in some Louisiana marshes (DeLaune et al., 1983a, Chapter 7). Vertical

accretion since 1963 was estimated with the ^{137}Cs dating technique (DeLaune et al., 1978). Two 50-cm long, 15-cm diameter cores were collected from unbroken marsh at the 125-m station of one transect. Two cores were also collected approximately 60 m from the 5-m station of the same transect towards the center of the broken marsh area, thus these cores were collected from the interior of the broken marsh. All cores were sectioned into 3-cm increments, dried and weighed; bulk density was estimated from the dry weight and wet volume of increments (DeLaune et al., 1983a). After determining ^{137}Cs activity in each increment, increments were ground, and subsamples were ignited at 400 ^{o}C to determine organic matter content (Davies 1974). Vertical accretion since was 1963 was determined by the depth of maximum ^{137}Cs activity.

As noted, broken marsh vegetation was expected to be more stressed than unbroken marsh vegetation, and several vegetational characteristics were used as indicators of stress. Stress is believed to reduce species diversity and increase the importance of a few common species (Odum, 1959:281). At Marsh Island, Nyman et al. (1993) noted that species diversity was greater and that Simpson's dominance index was lower in better drained marsh than in poorly drained marsh. Thus differences is species richness and Simpson's concentration index along transects were attributed to differences in degrees of stress along transects. Low above-ground biomass of important species was also considered indicative of soil waterlogging stress because soil waterlogging reduces end-season-standing crop above-ground biomass of emergent salt marsh vegetation (DeLaune et al., 1983b, Mendelssohn and McKee, 1988). These variables were analyzed as a completely randomized design with nesting of transect within area using Proc GLM of SAS (SAS

Institute Inc., Cary, North Carolina) to test for differences among the stations, which were classified according to their distance from the pond edge in the broken marsh, i.e., 5 m, 35 m, and 125 m. Variation resulting from burn history and area was accounted for with blocking; blocks and treatment by block interaction terms were pooled when appropriate ($P > 0.2000$). Tukey's studentized range test was used to compare stations when appropriate. If the hypothesis that marsh loss was related to plant stress and mortality was to be supported, the 5-m station should have had the lowest above-ground biomass, lowest species richness, and highest Simpson's index (lowest species diversity).

An attempt was also made to identify factors limiting vegetation. Physical factors and vegetational characteristics were analyzed with Proc GLM of SAS to explain variation in vegetation among the stations resulting from stresses such as insufficient elevation, insufficient soil drainage, and insufficient soil mineral matter content. Variation in vegetational characteristics resulting from site and burn history were accounted for by blocking; blocks were pooled when appropriate ($P > 0.2000$). Soil Eh data were analyzed as a 2 by 2 factorial, distance class by soil depth class. The interaction term was pooled ($P > 0.2000$). In all analyses, type III sums of square and a critical limit of 0.05 were used.

RESULTS

It became apparent during low water levels that accompanied north winds 10 and 11 October that the broken marsh areas were not hydraulically isolated as initially believed. Drainage of broken marsh ponds occurred via small channels that connected the broken marsh interior to large bayous and

lakes. These were not new channels; one is evident in Figure 2, which was taken in 1957. Drainage after frontal passage was near complete despite the fact that there appeared to be only one such channel for each broken marsh area and that the channels were relatively small, about 3-4 m wide and about 1 m deep.

Elevation varied among stations ($F = 11.92$, 2 and 16 df, $P = 0.0007$). Elevation was 6 cm greater at the 125-m stations than at the 35-m and 5-m stations (Table 11). Thus the broken marsh was lower in elevation as expected. Soil drainage also varied among the stations as indicated by soil water levels when the ponds were drained ($F = 6.51$, 2 and 16 df, $P = 0.0085$; $F = 6.35$) However, the pattern was not as expected. Soil was able to drain an average of 4 cm below the surface in broken marsh, but no drainage occurred at the other stations (Table 11).

Soil characteristics also varied among stations. Soil bulk density ($F = 6.35$, 2 and 16 df, $P = 0.0093$), mineral matter density ($F = 4.87$, 2 and 16 df, $P = 0.0223$), and organic matter density ($F = 6.01$, 2 and 16 df, $P = 0.0113$) were significantly greater at the 125-m stations than at the 35-m stations, with intermediate values at the 5-m stations (Table 11).

Soil Eh differed with soil depth ($F = 27.31$, 1 and 20 df, $P = 0.0001$), and among distance classes ($F = 5.72$, 2 and 20 df, $P = 0.0109$). Soil Eh differed 139 mV between the 125-m stations and the 35-m stations, but soil Eh at the 5-m stations was intermediate (Table 11). The lowest Eh occurred at the 35-m stations where low elevation combined with a lack of soil drainage, and the highest Eh occurred at the 125-m stations where elevation was highest (Table 11). Eh was intermediate at the 5-m stations where low elevation was mitigated by soil drainage (Table 11).

Table 11. Soil characteristics on 9 transects beginning in 3 broken marsh areas, and extending into surrounding solid marsh, October 1990, Marsh Island, Louisiana.

characteristic	mean (standard error) distance from edge of broken marsh 5 m	35 m	125 m
elevation (cm)	a0.0 (n.a.†)	a-2.1 (1.8)	b6.0 (1.4)
bulk density (g/cm^3)	ab0.10 (0.01)	b0.08 (<0.01)	a0.11 (0.01)
% weight mineral	44.9 (2.6)	41.6 (2.4)	46.3 (2.0)
mineral density (g/cm^3)	ab0.045 (0.004)	b0.036 (0.003)	a0.051 (0.004)
organic density (g/cm^3)	ab0.053 (0.002)	b0.049 (0.001)	a0.059 (0.003)
drainage potential (cm)	a-4.0 (2.4)	b1.5 (1.5)	b1.8 (0.8)
high-water depth (cm)	18.1 (1.0)	20.2 (2.2)	12.1 (1.3)
Eh at -1 cm (mV)	119 (11)	-34 (68)	146 (45)
Eh at -5 cm (mV)	-120 (43)	-140 (14)	-42 (36)

characteristic means with different letters are significantly different as indicated by Tukey's studentized range test ($P > 0.05$).

† no estimate of variation among the 5-m stations is available because the elevation of the 5-m station was defined as 0 for each transect, and was the basis for comparison among stations within each transect.

S. patens and *S. olneyi* accounted for 79% and 15% of the live above-ground biomass, respectively. The only other species that averaged more than 20 g m^{-2} at any station were *Aster tenuifolius* and *Juncus roemerianus* (Table 12). Distance into unbroken marsh did not account for significant variation in live *S. patens* above-ground biomass ($F = 3.34$, 2 and 16 df, $P = 0.0612$) even though above-ground biomass estimates at the 3 stations differed by as much as 200 g m-2 (Table 12). Nor did distance account for significant variation in the above-ground biomass of any other plant species, species richness, or Simpson's dominance index.

Variation in measured physical variables did not explain variation in total above-ground biomass and total live above-ground biomass. However, species richness was related to soil drainage ($F = 4.73$, 1 and 18 df, $P = 0.0432$) as well as to the interaction between burn history and soil bulk density ($F = 6.43$, 1 and 18 df, $P = 0.0207$). Interpreting these relationships was difficult, but it appeared that species richness increased as soil drainage and soil bulk density increased, and in response to burning. Species diversity appeared to increase from broken to unbroken marsh, as indicated by Simpson's dominance index (Table 12). However, no variables were significant even though soil drainage was nearly significant ($F = 4.21$, 1 and 14 df, $P = 0.0593$).

S. olneyi above-ground biomass decreased as soil drainage increased (Figure 23) ($F = 6.79$, 1 and 13 df, $P = 0.0218$). Above-ground biomass of live *S. patens* was greatest where soil bulk density was greatest, and was lowest where soil bulk density was <0.08 g/cc (Figure 24) ($F = 7.37$, 1 and 19 df, $P = 0.0137$). The interaction between elevation and soil drainage was also

Table 12. Live above-ground biomass of marsh vegetation (g/m^2), species richness (number of species/0.5m^2 plot), and Simpson's index (which increases as species diversity decreases) on 9 transects beginning in 3 broken marsh areas, and extending into surrounding solid marsh, October 1990, Marsh Island, Louisiana.

	mean (standard error) distance from edge of broken marsh		
	5 m	35 m	125 m
Spartina patens (Aiton) Muhl.	1,096.8 (94.6)	831.4 (73.9)	1,013.1 (82.5)
Scirpus olneyi Gray	204.8 (53.1)	287.8 (62.2)	259.2 (61.2)
Aster tenuifolius L.	88.2 (57.2)	62.0 (46.9)	18.3 (11.1)
Distichlis spicata (L.) Greene	12.7 (8.5)	11.2 (8.8)	1.5 (1.0)
Sporabolus spp.	4.8 (3.3)	10.9 (10.9)	3.2 (2.4)
Aster subulatus Michaux	4.6 (3.9)	2.0 (2.0)	0.5 (0.5)
Juncus roemerianus Scheele	0.7 (0.6)	0.0 (0.0)	41.3 (33.5)
Cyperus spp.	0.1 (0.1)	0.0 (0.0)	0.0 (0.0)
Fimbristylis castanea (L.) Vahl.	0.0 (0.0)	2.9 (2.8)	9.3 (5.8)
Pluchea camphorata (L.) DC	0.0 (0.0)	0.0 (0.0)	1.2 (0.8)
live all species	1,412.6 (105.9)	1,208.2 (104.5)	1,347.6 (67.7)
dead all species	564.5 (131.8)	477.0 (146.3)	690.2 (215.7)
richness (0.5 m-2)	3.7 (0.5)	3.0 (0.6)	4.0 (0.5)
Simpson's index	0.72 (0.05)	0.70 (0.04)	0.69 (0.04)

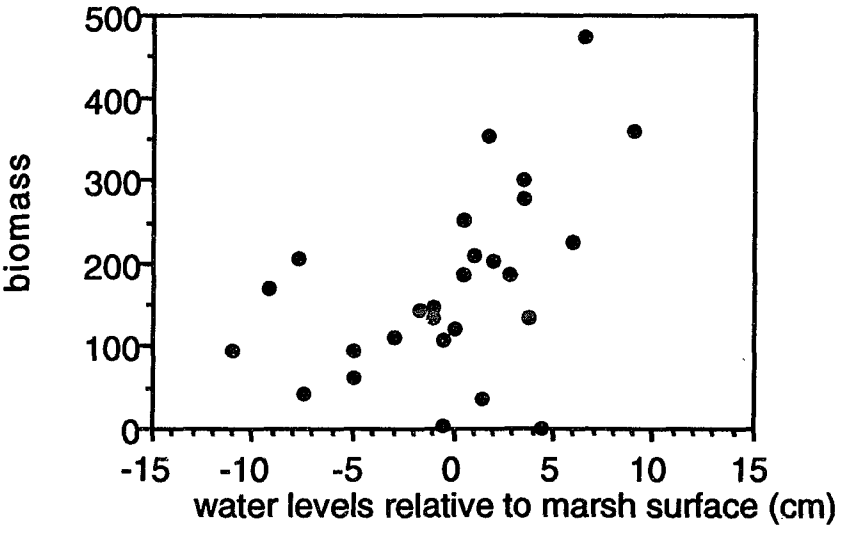

Figure 24. Relationship between biomass of *Scirpus olneyi* (g/m^2) and soil drainage, Marsh Island, Louisiana, October 1990.

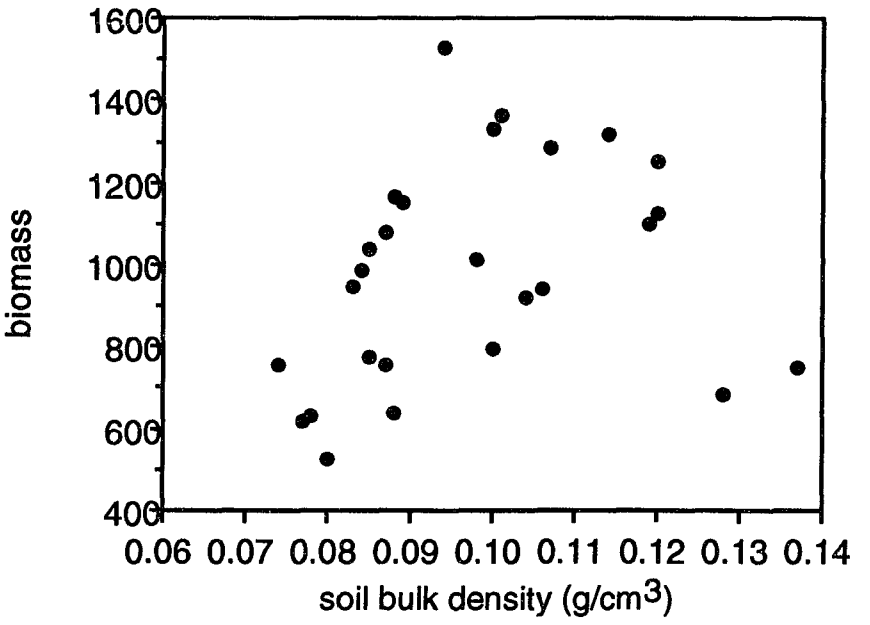

Figure 25. Relationship between biomass of *Spartina patens* (g/m^2) and soil bulk density (g/cm^3), Marsh Island, Louisiana, October 1990.

significant (F = 5.91, 1 and 19 df, P = 0.0251), and live *S. patens* above-ground biomass was lower where both were low.

Vertical accretion since 1963 was greater in the broken marsh interior than at the unbroken marsh site. Vertical accretion was 0.50 cm yr^{-1} in both cores from unbroken marsh, and 0.53 yr^{-1} and 0.67 cm yr^{-1} in the 2 cores from broken marsh. Bulk density profiles indicated that mineral matter accumulation in the broken marsh interior was apparently greater than that at the unbroken marsh sites for quite some time. Clay lenses were noted in the 49-51 cm increments of the 2 cores collected from the broken marsh area. Peat resumed below this lens, but bulk density could not be determined because the soil increment below 51 cm was incompletely sampled. Thus this lens does not represent a former lake or bayou bottom, and was more likely deposited by a major flood event. No such lens was found in the 2 cores collected from unbroken marsh. Either the flood event that deposited the lens at the broken marsh site did not deposit a similar lens at the unbroken marsh site, or the lens was further down in the soil profile at the unbroken marsh site. If the latter was the case, then the vertical accretion was greater at the unbroken marsh site than at the broken marsh site prior to 1963, even though vertical accretion was greater at the broken site since 1963. It could not be determined from these data which was the case.

DISCUSSION

Test of hypothesis

No difference was detected between broken marsh and unbroken marsh in vegetation vigor as indicated by live above-ground biomass of important species, species richness and species diversity. If I incorrectly failed

to reject the hypothesis that these means did not differ among distance classes, i.e., made a type II error, then *S. patens* above-ground biomass was actually greatest adjacent to broken marsh, which would also indicate that broken marsh vegetation was not more stressed than unbroken marsh vegetation. Soil Eh indicated that all sites could be classified as reduced (Patrick and DeLaune 1977), which is prevalent in wetland soils (Mitsch and Gosselink 1984:88). However, the pattern of soil Eh among sites indicated that broken marsh was not more waterlogging stressed than unbroken marsh as initially suspected. Nor was the broken marsh soil less drained as initially suspected. The hypothesis that marsh loss was related to vegetation stress was rejected for these reasons.

Above-ground biomass estimates were similar to other *S. patens* marshes in coastal Louisiana. White et al., (1978) reported that *S. patens* live above-ground biomass peaked in May near 1,300 g m-2. Hopkinson et al., (1978) also reported that *S. patens* live above-ground biomass peaked near 1,000 g m-2 one year, and near 1,400 g m-2 the next year. It therefore appeared that vegetation sampled at Marsh Island was not more stressed than in those other marshes, which supported the conclusion that marsh loss in these study areas was unrelated to vegetation stress. This contrasted sharply with the Lake Barre area marshes (Chapter 6 and 7).

Live below-ground biomass was not sampled, and strongly reducing conditions limit root growth by *S. patens* (Pezeshki et al. 1992) It is therefore possible that broken marsh contained less live below-ground biomass even though none of the data collected suggested that broken marsh vegetation was more stressed than unbroken marsh vegetation

Factors related to plant growth

Several workers have demonstrated the role of soil waterlogging in controlling *S. patens* growth (Bandyopadhyay et al. 1993, Burdick et al., 1989, Pezeshki and DeLaune 1990). Thus, it was not surprising that soil drainage was positively associated with above-ground biomass of *S. patens* and with species richness. What was surprising was that the greatest soil drainage occurred in broken marsh. Drainage of broken marsh soil did however agree with limited observations made during the pilot study (Chapter 8). Thus, these broken marsh areas appear similar, but on a smaller scale, to marshes where streamside sites are better drained than more inland sites even though the inland sites are more elevated (e.g., Agosta 1985, Redfield 1972). Such situations apparently result when there is little horizontal water movement in the soil (e.g., Hemond and Fifield 1982).

Scirpus olneyi is an important wildlife food plant, and habitat managers often attempt to manipulate environmental conditions to favor its growth (Chabreck et al., 1989). *S. olneyi* above-ground biomass was negatively associated with soil drainage, which may seem surprising. But this agreed with the finding of Palmisano (1967) that natural stands did not occur above minimum water levels. Such zonation may be explained by restriction of *S. olneyi* to less drained areas by competitive displacement with *S. patens*, as is the case between *S. patens* and *S. alterniflora* (Bertness, 1991). While that may occur, evidence also indicates that *S. olneyi* actually grows better when flooded. Chabreck and Narcisse (1981) found in a greenhouse study that *S. olneyi* culm density and length was greater in flooded soils than in drained soils.

Although I am unaware of previous documentation of a relationship between *S. patens* biomass and soil bulk density, such a relationship in these peaty soils was not surprising. Bulk density is a function of the mineral matter content in these peaty soils (Hatton et al., 1983), and mineral nutrients are important to marsh vegetation. Working on the Atlantic coast, Broome et al., (1975) found that the P concentration of the soil was related with above-ground biomass of *S. alterniflora*. King et al. (1982) reported that productivity of *S. alterniflora* was related to the soil Fe content, likely because Fe can neutralize H_2S toxicity. Above-ground biomass of *S. alterniflora* was also related to the soil content of P, K, Fe, Mn, and thus soil bulk density in Louisiana (DeLaune and Pezeshki, 1988, DeLaune et al., 1979). These data indicated that mineral sediments are also important to the health of *S. patens* marsh. Given a vertical accretion rate of 0.53 cm yr^{-1} and bulk density data in Table 1, differences in mineral matter accumulation of only 79 g m^{-2} exist between marsh with soil bulk density of 0.08 g/cm^3 and those with bulk density of 0.11 g/cm^3, even though they differ substantially in *S. patens* above-ground biomass.

As noted, it was surprising to learn that the ponds in the broken marsh areas were hydraulically linked to major water bodies. This is important because marsh soil generally drains only when it is adjacent to drained water bodies (Agosta, 1985, Gardner, 1975, Hemond and Fifield, 1982). Thus, soil drainage occurred in the broken marsh areas where it was least expected. Soil drainage was observed 5 m from tidally influenced water bodies in this marsh, whereas Gardner (1972) and Agosta (1985) reported little soil drainage 3 m and 4 m beyond salt marsh creekbanks on the United States Atlantic coast. This may partially reflect the longer period of low water in this study

area, over 48 hours, relative to that in response to low tides; i.e., 7 hours as reported by Agosta (1985).

Sedimentation

In addition to allowing significant soil drainage in broken marsh, the hydraulic linkage between the broken marsh and the large bayous and lakes allowed delivery of mineral matter to the broken marsh areas. However, this mineral matter did not penetrate far from the interior water bodies as indicated by the greater soil bulk density at the 5-m station than at the 35-m stations. This likely resulted from the widely recognized phenomena that most of the sediment load in flood waters settle out within a few meters of entering a marsh (e.g., Stoddart et al. 1989). The greatest soil bulk density occurred in the center of the broken marsh area as indicated by the cores taken for ^{137}Cs dating and by the bulk density of soil collected from the center of a broken marsh area during the pilot study (Chapter 8). The greater soil bulk density in the broken marsh interior than at the 5-m stations most likely resulted from the center of the broken marsh areas being adjacent to ponds longer than the marsh at the edge of broken marsh area, which was farther from these interior water bodies prior to the ongoing marsh loss.

The soil bulk density and mineral matter content at the 125-m stations indicated that these stations also received significant mineral matter. Even though these stations were the farthest from any water body, they were the closest to large lakes and bayous. Significant mineral sediments at the 125-m stations suggested that major flood events, which flood all elevations unlike tides, transported significant mineral sediments and distributed them more uniformly than tidal events. This agreed with previous work in Louisiana

marshes that demonstrated that storm events were more important than tidal events at promoting mineral matter accumulation (Reed, 1989).

Vertical accretion at Marsh Island, 0.53 cm yr^{-1}, appeared adequate to counter subsidence, 0.07 cm yr^{-1} (Coleman, 1966), and sea level rise, 0.24 cm yr^{-1} (Peltier and Tushingham, 1989). Greater vertical accretion in the broken marsh interior than at the unbroken marsh site likely resulted from the lower elevation. It is important to note that the broken marsh areas are where the marsh loss occurs (Nyman et al., 1990) even though they received the most mineral sediments and had a higher vertical accretion rate.

Mechanism of marsh loss

Even though no difference in vegetation was detected between broken and unbroken marsh, successive aerial photographs indicated that broken marsh converted to open water and expanded at the expense of surrounding unbroken marsh (Nyman et al., 1990). Rather than being related to vegetation stress, the mechanism of marsh loss appeared to be erosion of the unconsolidated soil below the living root zone at the marsh/water interface in the broken marsh areas. The roughly 25-cm thick living root layer of soil was suspended out over the pond areas for 30-40 cm in some places (Figure 26). At other places, such overhangs had detached from the adjacent marsh and come to rest on the pond bottom. Despite the fact that I have worked extensively at Marsh Island, I was unaware that the marsh water interface was undercut until I was caught in the middle of field work by the extremely low water levels produced by the strong and persistent north winds accompanying the winter weather front (I generally avoid sampling during such weather). This internal marsh erosion appeared similar to a description of marsh loss in another brackish marsh at Chesapeake Bay (Stevenson et al., 1985). Singh

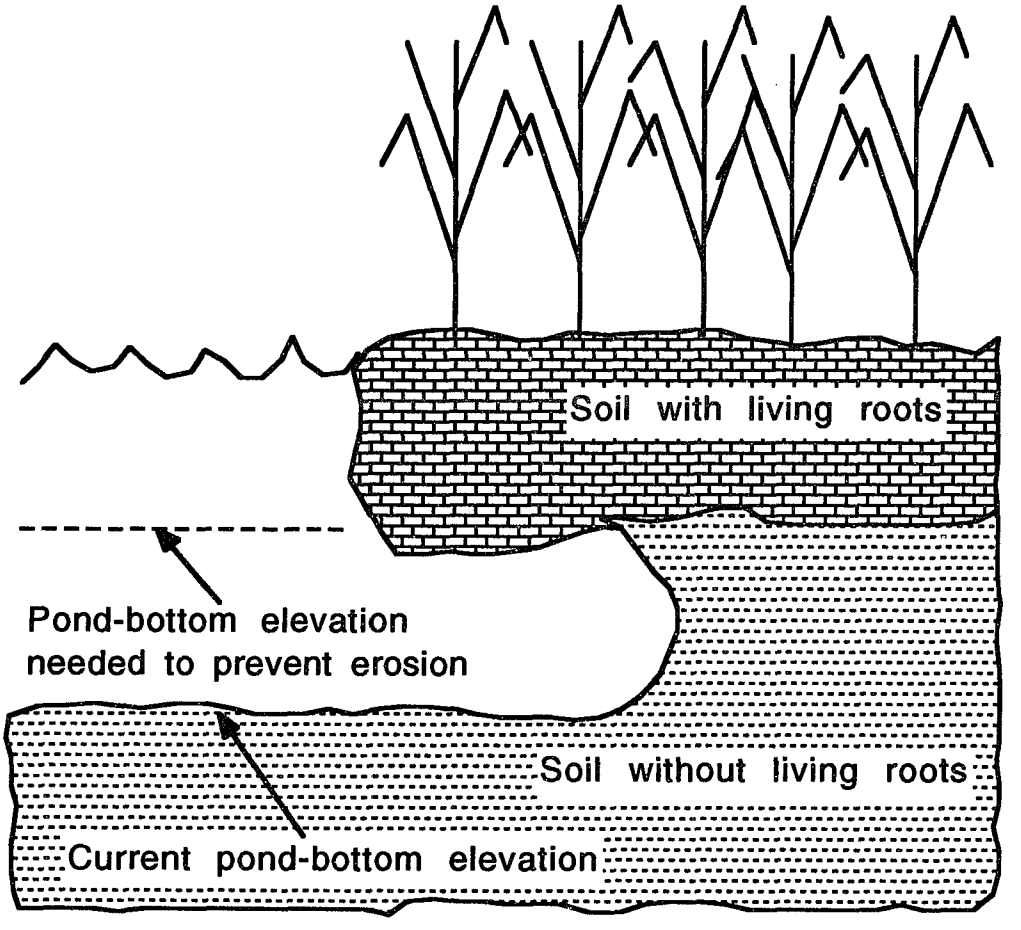

Figure 26. Illustration of the marsh water interface in the broken marsh interior at Marsh Island, Louisiana where marsh loss occurs. The living root mat appeared healthy and 15-30 cm thick. The pond bottom appeared to extend 10-20 of soil below the living root.

and Nathan (1965) also noted that the living root zone important to soil structure and strength; they stated that ditches cut below the root system caused cave-in of the ditch, whereas ditches that did not extend below the living root zone were stable. This mechanism of marsh loss is different from what was observed at the Lake Barre Marshes (Chapters 6 and 7). It therefore appears that marsh loss mechanisms vary spatially even within this region.

Erosion at Marsh Island evidently was not likely the result of tidal action. Weir-management, which reduces tidal exchange, did not reduce marsh loss on the eastern portion of Marsh Island, where marsh loss also occurred via the expansion of broken marsh (Nyman et al., 1990). This erosion may partly result from the drainage of storm tides. Harris and Chabreck (1958) noted that ponds enlarged and coalesced at Marsh Island following Hurricane Audrey. But it is also possible that mere exposure to pond water is enough to suspend the highly organic, structurally weak soil below the living root zone, which would also account for the similar marsh loss rates in weir-managed and unmanaged marsh. Outflow of soil water from under the root mat into the ponds during low water levels that accompany cold fronts might promote erosion and may occur even in weir-managed ponds that do not completely drain following cold fronts.

Marsh loss has probably proceeded via erosion rather than plant stress in these areas since at least 1957 because aerial photographs indicate that the broken marsh areas were hydraulically linked to the larger water bodies since at least then. However, plant stress and resulting dieback can be the marsh loss mechanism elsewhere, as demonstrated by the study in Lake Barre 135 km to the east (Chapters 6 and 7). Furthermore, plant stress may

have initiated the original ponds that have since spread via erosion at Marsh Island.

I hypothesize the following chain of events related to marsh loss on Marsh Island. Much earlier, all the marsh between the major bayous and lakes was unbroken marsh. Perhaps the bayous and large lakes were ringed by small natural levees that restricted drainage of rain and tide waters, which lead to plant mortality during a particularly wet year (flooding stress) or dry year (salt burns). It is also possible that small ponds were created by geese, muskrats, peat fires, alligators, and human activity (Lynch et al. 1947). Regardless of the cause, the elevation of the marsh surface decreased in areas where plant mortality was total because the living root network collapsed as has been observed at the Lake Barre area (unpublished data, R.D. DeLaune). This caused shallow pond area to replace marsh without any erosion of sediments.

These shallow ponds would not expand via erosion however, because the pond depth would not extend below the living root zone in the adjacent marsh. Instead, the ponds would begin to revegetate from the edges. Meanwhile, the surrounding marsh surfaces would continue accreting, as would the pond bottoms. However, marsh surfaces might accrete faster than pond bottoms (DeLaune et al. 1978). Thus, the pond bottoms would deepen relative to the marsh surface over time. Given enough time, the pond bottoms would become deeper than the living root zone of the adjacent marsh, which would cause the marsh edges to erode. The eroded marsh soil would quickly lose its structure, and only the 5-10% of soil volume occupied by mineral matter (Chapter 3) would be available to fill in the ponds. The ponds would therefore not likely fill in with eroded material. This hypothetical process may

partially explain why transient ponds observed by Turner and Rao (1990b) were all <20 ha; smaller ponds revegetate from the sides before they become deep enough to erode the unconsolidated soil below the living root zones in the adjacent marsh. Although this hypothesis requires formal testing, Lynch et al. (1947) noted that shallow creveys (ponds caused by muskrat eatouts) could be covered by spreading vegetation from the sides, but that deeper creveys caused tufts of marsh plants to become top-heavy and be washed out.

CONCLUSIONS

Soil Eh, vegetation above-ground biomass, and species diversity did not differ between broken marsh and unbroken marsh, and above-ground biomass was similar to that reported from other marshes. Thus, vegetation was no more stressed in broken marsh than in adjacent unbroken marsh and differences in plant stress alone did not explain differences in marsh stability. However, plant stress was involved on another scale because stress presumably limits the live root zone to the upper 20-30 centimeters of soil, which allowed the deeper soil to erode at the pond edges.

Two factors were related to spatial patterns in plant biomass: soil drainage and soil bulk density. Surprisingly, significant soil drainage occurred in broken marsh but not in unbroken marsh. Above-ground biomass of the dominant plant, *Spartina patens*, was lowest where soil bulk density was <0.08 g/cm^3, which illustrated the importance of mineral matter accumulation in submerging coastal marshes. The mechanism of marsh loss appeared to be erosion below the living root zone, as indicated by the vertical and often undercut marsh water interface, and by the separation of sod clasts. This is

different from more rapid marsh loss associated with plant stress observed in the previous studies at the Point Barre area (Chapters 6 and 7).

Studies to field check the hypothesis that interior marsh breakup is associated with vegetation stress need to be designed carefully. If an airboat was not available, the transects would have initiated in unbroken marsh adjacent to navigable bayous, and extended in the direction of broken marsh without being able to proceed completely to the edge of broken marsh. A gradient of decreasing above-ground biomass and elevation would have been noted, which would have been extrapolated to incorrectly concluded that marsh breakup was a function of inadequate elevation and resulting waterlogging stress on vegetation.

The internal erosion observed appeared to be a consequence of pond bottoms being deeper than the bottom of the living root zone in adjacent marsh, which suggests that the pond bottoms had a vertical accretion deficit rather than the marsh surface. This may be because the suspended sediments in coastal Louisiana are primarily clays, which require drying to become consolidated, and the ponds drain infrequently in these microtidal marshes.

SECTION V: CONCLUSIONS

CHAPTER 11
MANAGEMENT IMPLICATIONS AND FUTURE RESEARCH

LARGE SCALE MANAGEMENT IMPLICATIONS

This concluding chapter attempts to integrate the findings of the previous sections and put those findings into context with previous knowledge and mitigation efforts in coastal Louisiana. It is important to remember that one of the fundamental reasons that marsh loss exceeds marsh creation in Louisiana is that the marsh creation processes are not functioning at extremely low rates. This is partly a result of society's current methods for satisfying flood control, energy and navigation requirements. For at least 20 years, it has been recognized that net loss will continue as long as management of the Mississippi River does not change (Gagliano and van Beek 1973). Allowing greater flow down the Atchafalaya River or diversion of the Mississippi River onto the shallow continental shelf would speed marsh initiation. Once enough sediments have been deposited in those shallow bays to become exposed at low tide, then marsh vegetation could become established, create new marshes, and begin building up peat to counter sediment compaction. Increased sediments input into Atchafalaya Bay would also enhance marsh creation on the Chenier Plain. Once created, the marshes would require much less sediments for maintenance than were required for creation because the marshes accrete primarily by peat production. River diversion would also promote marsh maintenance processes by restoring spring flooding to some areas (Templet and Meyer-Arendt 1988). Although Louisiana marsh soils do not accrete directly by mineral buildup, mineral sediments provide plant nutrients that enhance peat

buildup. Thus, increasing nutrients might increase the upper limit to vertical accretion.

River diversion should also benefit marsh vertical accretion in areas affected by salt-water intrusion because the amount of soil mineral matter required for accretion is greater in the saltier marsh types. As noted in Chapter 3, this is probably related to sulfide toxicity rather than salinity per se. River diversion could prevent brackish marsh from converting to saline marsh, which might reduce wetland loss where brackish marsh vegetation dies as a result of salinity increases and the establishment of saline marsh vegetation is prohibited by a combination of low soil bulk density and high sulfate supply. Results from the study of CO_2 emissions and soil Eh suggested that such diversions might also increase soil Eh and lower soil respiration, thereby promoting peat production and accumulation. Perhaps freshwater diversion to convert saline marsh to brackish marsh will also prevent marsh loss by lowering sulfate supply and salinity, and allowing the establishment of *Spartina patens* in low density soils that induce *Spartina alterniflora* mortality. The difficulty will be differentiating between areas where salt-water intrusion is occurring because of wetland loss, and areas where wetland loss is occurring because of salt-water intrusion.

SMALL SCALE MANAGEMENT IMPLICATIONS: SOIL PROCESSES IN MANAGED MARSHES

Introduction

Unfortunately, wide spread river diversions will not likely be attempted because the benefits are not widely recognized or are considered to be less than the social and economic costs of diversions. Instead, much effort will

likely continue on a small scale effort. Many private landowners and governmental wildlife agencies manage coastal wetlands to improve wildlife habitat by producing favorable plant communities (Chabreck 1976). Some Louisiana marshes are managed to counter marsh loss by prevented salt-water intrusion (Berry and Voisin 1989), but none appear to be managed to counter rapid submergence as well. This management generally utilizes impoundments to manage water levels and will likely be preferred until there are widespread diversions from the river. Given the widespread practice, marsh managers should understand how soil formation may be influenced by marsh management practices. Thus, the purpose of this discussion is to review the relevant physical, chemical, and biological processes studied in this dissertation and put them into context with the larger, pre-existing body of knowledge. It is not my intent to encourage the impoundment of all marsh area. That decision involves many concerns. For instance, coastal marshes are important nursery areas to some marine fisheries, and marsh impoundments will reduce dependent fish populations where they are limited by the availability of nursery area.

Soil Formation

The amounts of mineral and organic matter in soil are not the same in the different marsh types (Templet and Meyer-Arendt 1988, Chapter 3). However, the amounts mineral and organic matter in marsh soil are related to each other because sedimentation supplies nutrients that promote plant growth and because vegetation promotes sedimentation by reducing water velocity and by biologically trapping sediments that are too fine to settle from the water column (Chapter 3). Thus plant biomass and soil mineral matter content are positively related in Louisiana brackish marsh (Chapter 10) and

saline marsh (DeLaune and Pezeshki 1988). Vertical accretion is determined by mineral matter accumulation in the southeast Atlantic coast and San Francisco Bay (see Stevenson et al. 1988), but by root growth in New England type marsh (McCaffrey and Thomson 1980, Bricker-Urso 1989) and Louisiana (Hatton et al. 1983, Chapter 7).

Mineral Matter Accumulation- Mineral sediments originate either from oceanic sediments carried landward by bottom currents such as in some southeastern Atlantic coast marshes (Meade 1969), from river sediments deposited directly by rivers such as in the active Mississippi River delta (Frazier 1967), or from river sediments delivered by marine processes such as in the Chenier Plain of Louisiana (Gould and McFarlan 1959). Different flooding events, whether river, tides or storms, differ in sediment delivery because sediment availability varies seasonally and annually. The timing of maximum sediment availability in coastal waters depends on discharge of nearby rivers. During low discharge years, sediment maxima occur shortly before or coincide with maximum river discharge, thus highest sediment availability occurs in late spring (Mossa and Roberts 1990). During high discharge years however, sediment maxima occur in winter or early spring when the river bed is scoured to accommodate increased flow (Mossa and Roberts 1990).

Tides deliver mineral sediments to southeastern Atlantic coast marsh, and thunderstorms can cause sediment losses (Stevenson et al. 1988). However, storms deliver more sediments than tides in Louisiana and New England type marsh (Stumpf 1983, Reed 1989). The process is well documented on the Louisiana coast (Reed 1989). Southeasterly winds preceding fronts increase water levels and wave action in bays and lakes. Wave action suspends bottom sediments from large water bodies, and high

water allows their transport to the marsh. Following frontal passage, prevailing northerly winds lower water levels and the marsh surface rapidly drains. Drainage allows the newly deposited sediments to dry and consolidate, which prevents their washing away during the next flooding event. This drying phase is especially important; Meeder (1987 cited in Reed 1989) noted that although Hurricane Juan deposited >6 cm of sediment, heavy rains following that hurricane washed away virtually all new sediment.

It may be possible to design and operate water control structures to compliment sedimentation processes, and still allow water level and salinity management. This would likely require that marshes flood before cold fronts and drain rapidly following their passage. However, these goals require such different practices that it may be most effective to manage for sediment capture every few years, and for wildlife habitat in other years (Cahoon 1991).

Organic Matter Accumulation- Organic matter accumulation depends on plant production, soil organic matter decomposition, and litter accumulation. Organic matter is extremely important in Louisiana marsh soils. It occupies more soil volume than mineral matter in fresh, intermediate, and brackish Louisiana marsh (Chapter 3). Organic matter accumulation, primarily via root growth, determines vertical accretion in Louisiana (Hatton et al. 1983, Chapter 7) and New England marshes (McCaffrey and Thomson 1980, Bricker-Urso 1989), and also reduces erosion (McCaffrey and Thomson 1980, and personal observation). Much is probably produced by perennials that produce more root biomass than annuals (Mitsch and Gosselink 1984:277). Peat production results from the production of adventitious roots just above the soil surface. Roots from one stem link up with roots from adjacent stems to form a tightly interwoven root network that forms the new, more elevated marsh surface.

This process depends on flooding that induces anoxia and hypoxia in the plants. Although death often results in the deeper parts of the plant where anoxia develops, ethylene production stimulates adventitious root growth in the hypoxic portions at the soil surface (Hook 1984, Kozlowski and Pallardy 1984, Jackson 1985).

Plant Growth- Marsh managers may be able to contribute greatly to soil formation by promoting vigorous plant growth. Wildlife habitat is usually improved by producing desirable plant communities (Chabreck 1976) but impoundment drawdowns increase growth as well (e.g. Carney and Chabreck 1977, Kadlec 1962). Although growth is stimulated by drainage, adventitious root production is not likely stimulated. Thus, current marsh management practices may have to be modified to allow more flooding than is optimal for plant growth to stimulate this vegetative growth process of marsh vertical accretion.

Draw-downs might also speed the decomposition of peat that took many years to accumulate, thus reducing the elevation of the marsh surface and increasing waterlogging stress on marsh vegetation upon reflooding. The different responses of CO_2 emissions to water-table depth among the marsh types (Chapter 4) may make fresh and saline marsh soils more susceptible than brackish marsh soils to such losses. Reduced water-table depths could result either from intentional drawdown of managed marshes or from the construction of canals and ditches. It is not clear how canals have affected water-table depths in Louisiana marshes, but elsewhere, canals and mosquito ditches have lowered the water table in some marshes, and increased flooding in others (see Balling and Resh 1983). Managed marshes are sometimes intentionally drawn down every few years to promote seed germination of

annuals, but it is not clear how water-table depths are affected in the marsh interior. However, drawdowns might not significantly affect peat decomposition given the small degree of soil drainage observed in interior Louisiana marshes (Chapters 9 and 10) and East coast marshes (Hemond and Fifield 1982, Agosta 1985).

Mineral sediment inputs promote plant growth because they contain phosphorus (DeLaune et al. 1981). Phosphorus may be a limiting plant nutrient in sediment poor marshes such as fresh marshes, but not in sediment rich marshes such as salt marshes (Patrick and DeLaune 1976). These are the most likely reasons that relationships have been observed between soil bulk density and plant biomass in saline marshes (DeLaune and Pezeshki 1988, Chapter 9) and in brackish marshes (Chapter 9).

Two additional, major factors limiting plant growth are related to the microbial decomposition of soil organic matter in waterlogged soils: soil sulfides and low soil Eh. Sulfides occur only when Eh is low but the reverse is not true, and low Eh alone is not as stressful as sulfides and low Eh (Pearson and Havill 1988, Pezeshki et al. 1988).

Soil Eh can be higher in brackish soil than in fresh and saline marsh even when water-table depths are the same (Chapter 4). This suggests that with similar degrees of soil drainage, less stress would result from low Eh on brackish marsh vegetation than on fresh and saline marsh vegetation. The most likely reason for this is that microbial respiration rates differ among fresh, brackish and saline marshes.

Sulfides, or reduced sulfur, also affect plant health. Although oxidized sulfur in the form of sulfate, SO_4^{2-}, is not toxic at natural concentrations, reduced sulfur, i.e., sulfides: H_2S, HS^-, or S^{2-}, are (Pearson and Havill 1988,

Pezeshki et al. 1988). The degree of sulfide stress on vegetation is not likely the same among fresh, brackish, and saline marsh. Sulfate reduction depends partly on SO_4^{2-} availability (Nedwell 1982), and SO_4^{2-} increases from fresh to brackish to saline marsh because seawater is the primary SO_4^{2-} source (Brupbacher et al. 1973, Feijtel et al. 1988). Thus sulfide stress is likely greater in saline marsh than in fresh marsh, which is the most likely reason that plant stress and mortality occurs in excessively flooded saline marsh soils (DeLaune et al. 1983b, Mendelssohn and McKee 1988), whereas near permanent flooding can be survived by many fresh and intermediate marsh plants such as species of *Scirpus*, *Zizaniopis, Cladium,* and *Typha.*

Soil Fe apparently buffers sulfide toxicity (King et al. 1982). Sulfate reducing bacteria become active only after Fe reducing bacteria exhaust the oxidized Fe pool. The resulting reduced iron pool is also important because reduced iron precipitates with reduced sulfur, thus removing sulfides from the soil solution. Saline marsh requires almost twice as much mineral matter as brackish marsh for soil formation (mineral and organic matter relationships section), likely to provide Fe for SO_4^{2-} buffering and precipitation (Chapter 3), and soil iron is positively associated with saline marsh plant production in Georgia marsh (King et al. 1982). Thus salt-water intrusion probably increases sulfide stress and sedimentation requirements as well as salinity stress.

Litter Accumulation- Litter accumulation might be important for accretion in some marshes, although examination of scores of shallow cores suggests that root production is most important in Louisiana (personal observation) and New England marsh (McCaffrey and Thomson 1980). Where litter is an important source of soil organic matter, tidal flushing may

affect organic matter accumulation. Generally, tidal flushing removes less litter from Texas and Louisiana marsh than from Atlantic coast marsh, reflecting differences in tidal range (Hopkinson et al. 1978). As expected, tidal flushing removes more litter from saline marsh than from brackish marsh (White et al. 1978). Also, detritivores such as fiddler crabs (*Uca* sp.) are apparently abundant enough to limit detrital buildup on the marsh floor in salt marshes.

Decomposition- Soil decomposition is continuous on the Gulf Coast and southeastern Atlantic coast even though rates are slower in winter than in summer (Morris et al. 1986, Smith et al. 1983). This oxidizes significant amounts of soil organic matter that must be replaced to maintain marsh elevation Soil drainage increases decomposition (Chapter 4), thus excessive drainage could oxidize a great deal of soil organic matter. Organic matter requirements for marsh soil formation differ among marsh types because brackish marsh soil decomposes slower than fresh and saline marsh soils. Thus, plants resistant to decomposition may promote soil formation. All brackish marsh species may not decompose at the slow rate observed for *Spartina patens* soil in Chapter 4, and it is possible that other plant species may decompose at different rates. If other brackish plant species provide a better diet for soil microbes, then other brackish marshes may have faster soil decomposition rates and lower soil Eh than that observed in Chapter 4.

Recommendations

If the rate of global sea level rise increases as expected, it will be accompanied by salt-water intrusion (Titus 1986). Options designed to counter these processes are currently needed in managed marshes in some areas of Texas and Louisiana, and may be needed elsewhere in the near

future. It would be a mistake to focus management solely on prevention of salt-water intrusion, which would likely result in practices that limit soil formation, or solely on sedimentation processes, which would likely result in practices that do little to maintain or improve wildlife habitat. Thus an integrated approach to simultaneously counter these processes in managed marshes is needed. It should be noted that some degrees of submergence and salt-water intrusion may be so great that it would be impractical to prevent marsh loss.

Promoting vigorous plant growth and the vegetative growth mechanism of marsh vertical accretion appears to be the best counter-measure to rapid submergence and salt-water intrusion. As noted, flooding is required to stimulate the production of adventitious roots. Vigorous plant growth requires optimum salinity and water levels, and a certain amount of mineral sediments. Obviously, optimum water levels cannot be maintained at the same time that peat production is promoted.

Increasing sedimentation may also improve plant growth on mineral poor soils. Inadequate sedimentation might contribute to inadequate plant growth and subsequent inadequate organic matter accumulation, vertical accretion deficits and wetland loss (Chapters 6 and 7). Rapidly submerging, sediment poor marshes should therefore be flooded frequently with sediment rich water, as well as rapidly drained so that newly deposited sediments consolidate before the next flooding event. Thus management practices should incorporate knowledge of river discharge and the relative importance of tides or storms in the sediment delivery processes. For instance, low natural levees that are topped by storm tides, and large-capacity water control structures that allow rapid drainage during the low water levels that follow cold

fronts might be successful management tools in some areas of Louisiana where litter accumulation in unimportant.

Preventing brackish marsh from converting to saline marsh during salt-water intrusion might reduce wetland loss, especially in sediment poor areas. Brackish conditions reduce SO_4^{2-} influx, and thus should reduce sulfide stress on vegetation, while *Spartina patens* soil organic matter decomposes slowly, reducing soil organic matter losses as well as waterlogging stresses on vegetation. This should also prevent mineral matter requirements from increasing as brackish marsh converts to saline marsh. Such measures may not be necessary where salt water intrusion is accompanied by increased sedimentation sufficient to counter submergence of saline marsh.

Maintenance of fresher marsh in more saline areas via impoundments is not without risk. At some point, the benefits of drawdown to plant growth will be outweighed by increases in soil organic matter decomposition. While there is not enough information with which to determine how long of a drawdown is too long, fresh marsh impoundments likely cannot tolerate as lengthy of a drawdown as brackish marsh impoundments because soil organic matter decomposes so much faster in fresh than in brackish marsh. Furthermore, if impoundment levees fail, extensive salt burns likely follow. Even more important than actual plant mortality, is the collapse of surface peats (over 10 cm) following plant mortality (DeLaune et al., in review). It may be near impossible to reestablish plant growth in resulting depressions before soil erosion begins unless forced drainage is available. This may be especially important if management had previously negatively affected soil formation processes, which might not be apparent in functional impoundments because of water level control.

Hopefully, more will be learned about wetland soil processes and the effects of various management strategies. Marsh managers may use the following types of data to help evaluate the magnitude of submergence in their region, and the effects of marsh management on soil formation. Submergence rates can be estimated from long term, i.e. 20-50 years, tide gauge analyses. The U.S. Army Corps of Engineers and the National Ocean Survey are two data sources, and those data from the Gulf of Mexico are already analyzed and published (Ramsey and Penland 1989). Soil formation can be estimated by marking the marsh surface with a layer of feldspar (Cahoon and Turner 1990). Filter papers techniques can indicate the relative contributions of tide and storm events (Reed 1989). A record of weather conditions, operation of water control structures, continuous water level data, and soil formation will likely also be necessary to evaluate the factors controlling soil formation in managed marsh.

RECOMMENDATIONS FOR FUTURE RESEARCH: ARE LANDSCAPE PATTERNS RELATED TO MARSH LOSS PROCESSES?

Vegetation stress followed by plant dieback and pond formation is believed to be the primary mechanism of marsh loss (Gagliano et al. 1981, Turner 1990). Plant stress is commonly assumed to originate from one of two sources in Louisiana. The earliest recognized plant stress was saltwater intrusion into non-saline marshes caused an absence of overbank flooding by the Mississippi River, and the presence of a network of canals throughout the marsh zone (Viosca 1928). The resulting conversion from less saline conditions to more saline conditions has been associated with rapid marsh

loss (Sasser et al. 1986). Later, it was recognized that rapid subsidence also contributes to marsh loss (Gagliano and van Beek 1973). In some marshes, vertical accretion is slower than submergence. Such marshes are slowly sinking lower and lower relative to mean water levels, which results in flooding stress on vegetation and subsequent marsh loss (DeLaune et al. 1983).

Leibowitz and Hill (1987) recently reported that marsh loss occurs in two landscape patterns in coastal Louisiana. Marsh loss was scattered throughout the marsh interior in the most common landscape pattern. Marsh loss rates in these areas averaged less than 0.5%/yr. Less common were areas where marsh loss was concentrated in large hotspots. Marsh loss rates in these areas averaged just over 2.7%/yr. Hotspots occupied less than 12% of all marsh, but accounted for almost 43% of all the marsh loss in that previous study (Leibowitz and Hill 1987).

The two field studies documented in this dissertation were conducted in a hotspot (Chapters 6 and 7) and a broken marsh landscape (Chapters 9 and 10). Those studies were designed as detailed examinations of marshloss processes, rather than as test of hypotheses regarding landscape patterns. Rigorous, statistically valid comparisons would require that more than one site be studied in each landscape pattern. However, comparing the findings of those 2 field studies may suggest avenues for such future research.

Plant production was lower in this study site than in other Louisiana marshes (Chapter 6). There was therefore less organic matter available for soil formation and for export to the surrounding estuary at this site than at other marshes. Aboveground, belowground biomass ratios and soil Eh indicated that flooding stress on vegetation was high. Thus the poor

production was attributed to flooding stress, as would be expected if there was a vertical accretion deficit.

Vertical accretion in the Lake Barre study area was extremely rapid relative to other marshes, which was unexpected. Vertical accretion averaged almost 1 cm/yr (Chapter 7), which was substantially greater than the average for southeast Louisiana, which is 0.72 cm/yr (Chapter 3). Although vertical accretion was extremely rapid, it was inadequate to counter submergence in the area. Submergence was estimated to be greater than 1.3 cm/yr (Chapter 2) . Thus, inadequate vertical accretion leading to excessive flooding stress on marsh vegetation, and subsequent plant die-back and pond formation was indicated as the marsh loss mechanism at this site.

Marsh loss could be countered in this study area with extremely large mineral sediment additions that do not seem feasible. At least 11 cm of mineral sediments would have to be pumped onto the marsh surface just to restore marsh elevation to normal. Assuming that such a mineral deposit would have a bulk density of 0.80 g/cm^3, then 88 kg/m^2 of sediment would be required. Furthermore, additional sediments would be needed each year to counter on-going submergence. The sediment requirements for brackish and saline marsh estimated in Chapter 3 and the amount of mineral sediments actually deposited in the study area were used to estimate the amount of mineral sediments required each year in addition to those naturally deposited. These calculations indicated that mineral sedimentation in saline marsh would have to increase by 0.5 kg m^{-2} yr^{-1} so that it was 2.5 g m^{-2} yr^{-1}. Mineral sedimentation in brackish marsh would have to increase even more, by 0.7 kg m^{-2} yr^{-1} so that it was 1.4 kg m^{-2} yr^{-1}.

Conditions were very different at Marsh Island where marshloss was scattered throughout the marsh interior (Chapter 10). Vertical accretion averaged 0.55 cm/yr, which was much slower than at the Lake Barre site. However, vertical accretion was adequate to counter the moderate submergence rate in this area, which was estimated at only 0.31 cm/yr. Vertical accretion was actually greater in broken marsh (0.60 cm/yr) where marsh loss occurred, than in solid marsh (0.50 cm/yr) where marsh loss did not occur. Contrary to initial expectations, there was no difference in soil Eh between solid marsh and broken marsh, and broken marsh soil was well drained. End-of-season, standing-crop plant biomass at this study site was typical of healthy brackish marshes, and did not differ between broken marsh and solid marsh. These data indicated that marsh loss at this site was unrelated to either salinity stress or flooding stress.

The mechanism of marsh loss appeared to be soil erosion below the living root zone, as indicated by the vertical and often undercut marsh water interface, and by the separation of sod clasts. I was unaware that the marsh water interface was undercut until we were caught in the middle of a field trip by a winter weather front that produced extremely low water levels. This also appears similar to the erosion of floating, fresh marsh in Louisiana described by Gagliano and Wicker (1989), except that erosion at Marsh Island does not seem to be related to tidal action. Thus, some marsh loss in Louisiana is not associated with plant stress as is currently believed, but is similar to the internal erosion reported in a Chesapeake Bay brackish marsh (Stevenson et al. 1985).

It was concluded that marsh loss in this study area could be countered only by increasing the elevation of the pond bottoms so that the loosely

consolidated soil below the living root zone at the marsh/water interface would not be exposed to open water (Figure 24). This may be possible without pumping sediments. If the broken marsh areas could be drained for a short time each year for several years, perhaps emergent vegetation could grow in the pond areas and build up the elevation of the pond bottoms by the production of a thick root mat. It might be possible to achieve this goal with modification of current Louisiana marsh management techniques, which are usually directed at improving wildlife habitat by producing favorable plant communities.

The difference mechanisms observed in the 2 study areas relate to differences in the strength and structure of organic matter in the soil. Soil strength and structure have been a recurring issue in these studies. Organic matter provides soil structure (Chapter 3), the structure can be either weak (Chapter 5) or strong (Chapter 10). Furthermore, part of this structure can collapse (Chapter 7), be eroded, or remain stable (Chapter 10). Refining our understanding of the properties that contribute strength to soil would likely improve our understanding of marsh loss processes.

It is important to determine why different processes are important in different areas. Internal erosion might be as widespread in Louisiana as the broken marsh interior landscape pattern is, or if hotspots might always result from plant stress. It is important to determine if this is so because landscape patterns could then be used to guide restoration efforts. It is also important to determine if internal erosion is widespread because marsh restoration efforts generally seeks to prevent marsh loss by preventing plant stress, but internal erosion occurs in marshes that are considered healthy.

Further study is also needed in areas where marsh loss is related to plant stress to determine the role of mineral sediments in reducing plant stress. Several field studies have noted positive relationships between plant biomass and soil nutrient content (Broome et al. 1975, DeLaune et al. 1979, DeLaune and Pezeshki 9188, King et al. 1982). Perhaps fertilizers could be applied when mineral sediments are not available. It might be critical to address nutrients before plant stress becomes severe because Wilsey et al. (1992) reported that fertilizers applied to a die-back zone in a saline marsh was not beneficial, which suggests that nutrients alone are of little value to marshes that have already begun to die.

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Vita

John Andrew Nyman was born 13 May 1960 in Hattiesburg, Mississippi but was raised in New Orleans, Louisiana where he attended grade school. He received a B.S. with a major in Biological Sciences from the University of New Orleans in December of 1984. From June 1985 through January 1987, he worked for the Louisiana Department of Wildlife and Fisheries as a Wildlife Specialist II. His duty station was the remote Pass A Loutre Wildlife Management Area that lies at the very end of the Mississippi River in the Bird's Foot Delta. In January 1987 he entered the Graduate School of Louisiana State University in Baton Rouge to work on a M.S. with a major in Wildlife under the direction of Dr. Robert Chabreck. In December of 1988, he took a full-time position as a Research Associate II at the Laboratory for Wetland Soils and Sediments, which is now known as the Wetland Biogeochemistry Institute. Since that time he married the former Cabrina Angel Bernardi, completed his M.S. degree, celebrated the birth of Lauren Elizabeth Nyman, and began working on a Ph.D. with a major in Oceanography and Coastal Sciences under the direction of Dr. William H. Patrick, Jr. He is currently a Research Associate III at the Wetland Biogeochemistry Institute and a candidate for a Ph.D.

DOCTORAL EXAMINATION AND DISSERTATION REPORT

Candidate: John A. Nyman

Major Field: Oceanography and Coastal Sciences

Title of Dissertation: Soil Processes Related to Marsh Loss in Coastal Louisiana

Approved:

Major Professor and Chairman

Dean of the Graduate School

EXAMINING COMMITTEE:

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